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13th Central Hardwood Forest Conference

April 1-3, 2002

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April 1-3, 2002

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13TH CENTRAL HARDWOOD FOREST CONFERENCE

Proceedings of a Conference
held at the
University of Illinois
Urbana-Champaign, Illinois

April 1-3, 2002

Edited by:
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Jeffrey O. Dawson, University of Illinois
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Sponsored by:
Department of Natural Resources and Environmental Sciences,
University of Illinois at Urbana-Champaign
Illinois Society of American Foresters
and
North Central Research Station, USDA Forest Service

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Jeffrey O. Dawson (program committee) and
Jerry Van Sambeek (proceedings editor)

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Felix Ponder, Jr.; Ed Loewenstein, and Jim Fralish

FOREWORD

This conference was the 13th in a series of biennial meetings that have been hosted by numerous universities and research stations of the USDA Forest Service in the Central Hardwood forest region in the eastern United States. The purpose of the Conference has remained the same since its inception—that is to provide a forum for the formal and informal exchange of information concerning the ecology, siliviculture, and management of the Central Hardwood forest by scientists, foresters, and practitioners throughout the region. As with previous conferences, a wide range of subjects was presented representing research efforts to understand this diverse forest resource. An especially noteworthy shift from previous conferences was the increase in oral and poster presentations related to forests in riparian zones and other wetlands and water quality issues. Altogether, there were 60 oral and 33 poster presentations accepted for this conference yielding these proceedings containing 58 peer-reviewed papers, 23 research notes, and 9 abstracts.

REVIEW PROCEDURES

Manuscripts from oral presentations were assigned to one of the editors and peer-reviewed by at least two professionals unless otherwise indicated. Reviews were returned to authors to revise their manuscripts and resubmitted electronically to the North Central Research Station, USDA Forest Service for final editing and publishing. The proceedings editor or an associate editor reviewed the research notes and abstracts. Authors are responsible for the accuracy and content of their papers.

LIST OF TECHNICAL REVIEWERS

The editors sincerely appreciate the efforts of the following colleagues who reviewed one or more manuscripts. Their efforts greatly improved the content of this conference proceeding.

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CONTINUING FORESTRY EDUCATION

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The Planning Committee wish to acknowledge the insightful welcoming messages from Dick Warner, UIUC, and Clark Baldwin, USDA Forest Service. We would also like to express our appreciation to Mary Beth Adams, Jeff Dawson, James Fralish, John Groninger, Louis Iverson, Ed Loewenstein, Mike Ostry, Steve Pallardy, Felix Ponder, John Taft, and Jerry Van Sambeek for serving as session moderators. A special thank you goes to the University of Illinois for providing the cover art. Finally, we need to acknowledge the exceptional work by Barbara Winters and others who helped format the manuscripts and assembled the proceedings and electronic media.

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THE CENTRAL HARDWOOD FOREST: ITS BOUNDARIES AND PHYSIOGRAPHIC PROVINCES

James S. Fralish[1]

ABSTRACT.—The Central Hardwood Forest (CHF) refers to the area where deciduous hardwood species overwhelmingly, but not exclusively, dominate the stands and cover types that occur as repeating units across the landscape. Transition zones where Central Hardwood species mix with species from adjacent regions identify boundaries of the region. These regions are the Northern Hardwood-Conifer Forest along the northern border, the Southeastern-Pine Forest along the eastern and southern borders, and the Tall Grass Prairie region to the west. There is a distinctness and cohesiveness to the CHF as its boundaries frequently cut across geographic features. The 18 oak and 10 hickory species that dominate stands from Missouri to West Virginia, and Wisconsin to Alabama, unify the region. The more important species such as white, black, and chestnut oak may form essentially climax communities on dry sites or successional communities on moist sites. These species may be regarded as obligate xerophytes and facultative pioneers. Such a successional/stability pattern/process is either absent or difficult to identify in other forested regions. Geographically, the region is also diverse. Physiographic provinces include the unglaciated Blue Ridge Mountains, Appalachian Plateaus, Interior Low Plateaus, and Ozark Plateaus, and the glaciated Central Lowlands. The Mississippi floodplain and Gulf Coastal Plain extend into the region. Bedrock, surface deposits, topography, and the soil mosaic vary from province to province and with subregions within provinces.

What and where is the Central Hardwood Forest region and are there boundaries that can appropriately define it? The answer to that question varies somewhat depending on which scientist, book, research report, or map is consulted (Fralish 2002). The concept of a "Central Hardwood Forest" (CHF) region was developed and used by foresters before E. Lucy Braun alluded to the region in 1950. The name apparently developed over the years from the idea that the vegetation of a relatively large area in east-central United States was dominated by broadleaf deciduous hardwoods (angiosperms) and that few commercially important conifers (gymnosperms) or evergreen angiosperms were found within the area. "Vegetation" here is defined as the sum of the stands (communities) or cover types in the region.

Using the criterion of defining the CHF by the dominance of hardwood stands and a general absence of commercial or non-commercial conifers is not completely satisfactory. The generally noncommercial redcedar (*Juniperus virginiana* L.) is the only conifer that spans the CHF, but its development as a community type is severely restricted. Limestone cedar glades are common in central Kentucky and Tennessee and a few sandstone cedar glades occur in the Shawnee Hills of southern Illinois. The species is invasive on open xeric sites such as prairies, barrens, and woodlands of Missouri and Arkansas. It may survive but seldom grows and develops (completes the life cycle) in moderately dense woods.

Other conifers in the CHF include shortleaf pine (*Pinus echinata* Mill.) and Virginia pine (*P. virginiana* Mill.) on xeric sites scattered in Kentucky and Tennessee, and the southern Appalachian Mountains. Shortleaf pine is found in only two locations in southern Illinois, scattered in the Ozark Mountains of Missouri, and as an important community type in the Boston Mountains of the Ozark Plateaus, the Coastal Plain, and the Piedmont province. The largest

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part of the range of Virginia pine is in the CHF. The ranges of several other important conifers (e.g., white pine, P. strobus L.; eastern hemlock, *Tsuga canadensis* (L.) Carr; and eastern larch, Larix laricina (DuRoi) K. Koch) extend into the region. Although pitch pine (*Pinus rigida* Mill.) has a relatively small range, it is confined to the CHF.

Therefore, the name "central hardwood," should be interpreted to mean that deciduous hardwood species overwhelmingly, but not exclusively, dominate the stands and cover types that occur as repeating units across the landscape. With this definition, it should be possible to identify the boundaries of the CHF by changes in forest composition and pattern of repeating units.

One common approach to defining the CHF is the reporting of forest statistics (e.g., volumes, net growth, etc.) by states, but political boundaries do not ecologically define the boundaries of a particular forest region. For example, in the proceedings of the sixth CHF Conference, Spencer and Bones (1987) reported forest statistics by state, thus their boundary for the CHF region followed state boundaries. They included all of the states of Missouri, Iowa, Illinois, Indiana, and Ohio, but the approach excluded southern New England states and parts of various other states such as southern Wisconsin, Michigan, and Minnesota that others (e.g., Braun 1950, Kuchler 1964, Leopold and others 1998) have included. The problem is that the operational areas of research experiment stations include only entire states, and some of these states often encompass parts of different natural forest regions (e.g., Central Hardwood and Northern Hardwood-Conifer Forest) (Lull 1968, Merz 1978, Kingsley 1985).

In contrast, the "central hardwood" region of Braun (1950) included five forested subregions called the Mixed Mesophytic, Western Mesophytic, Oak-Chestnut, Oak-Hickory, and Beech-Maple (fig. 1), all of which bisect one or more states. In the Proceedings of the First CHF Conference, Clark (1976) generally defines the CHF following Braun's regions, but excludes the oak-chestnut area. Hicks (1998) based his map on and considered the silviculture of forest types in four of the five regions: Mixed Mesophytic, Western Mesophytic, Oak-Chestnut, and Oak-Hickory (Ozark Plateau subregion only). He excluded the Beech-Maple region, the northern section of the Oak-Hickory region (the glaciated Central Lowland Physiographic province), and the Prairie-Forest Transition of the Southern Division of the Oak-Hickory Section.

A third interpretation of the CHF boundary is generally outlined by Leopold and others (1998), Patton (1997), and Fralish and Franklin (2002), but with small variations (fig. 2). The three maps include the five regions identified by Braun (1950). All authors extended the CHF boundary as far west as eastern Oklahoma, to near or into Kansas and Nebraska, as far north as central Minnesota, Wisconsin, Michigan, and into Ontario. The northern boundary extends from Minnesota to Cape Cod and Massachusetts in New England and includes a portion of southeastern New York. The eastern boundary includes the Appalachian Mountains and with the southern boundary across northern Georgia, Alabama, and Mississippi. There is general agreement that northern Pennsylvania and most of New York are excluded, the communities of these states being part of the Northern Hardwood-Conifer Forest (Fralish and Franklin 2002). The primary differences, albeit minor, are that Leopold and others (1998) extends the CHF north into the southern tip of Quebec which is well into the Northern Hardwood-Conifer Forest, and well beyond the CHF area on the maps of Farrar (1995) and Scott (1995). Patton (1997) extends the region south through the southern evergreen and semi-evergreen forest of central Texas and the Coastal Plain to near the Gulf Coast.

Given the variation in location of the CHF boundaries, the intent of this manuscript will be to

1) appropriately define criteria for delineating the CHF boundaries,
2) delineate the boundaries, and
3) briefly describe the physiographic provinces and subdivisions included within the CHF region.

CRITERIA FOR DELINEATING THE CHF BOUNDARY

Bailey (1996) reviewed the question of boundary criteria and the reliability of climate, vegetation, soil, and physiography in delineating ecosystems (forest regions, formations, or biomes). It is surprising that he found all of these approaches except macroclimate less than satisfactory and dismissed them rather quickly. However, the arguments used to discount these factors need closer examination. Bailey states "Vegetation and associated fauna, or biota, are constantly changing due to disturbance and

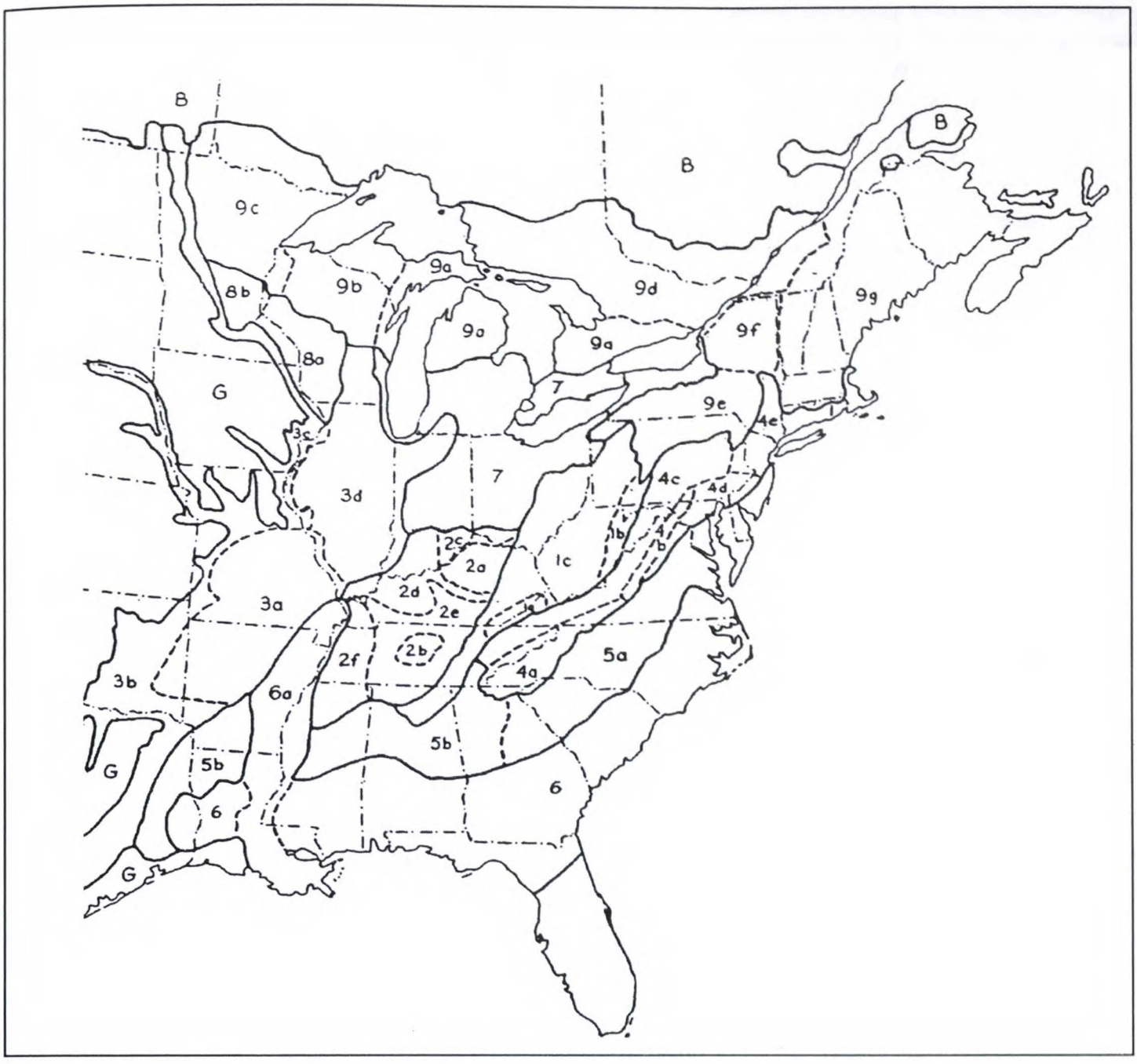

Figure 1.—*Forest and physiographic regions of eastern United States:*

(1) Mixed Mesophytic Region, a. Cumberland Mountains, b. Allegheny Mountains, c. Cumberland and Allegheny Plateaus;

(2) Western Mesophytic Region, a. Bluegrass, b. Nashville Basin, c. Area of Illinoian Glaciation, d. Hill Section, e. Mississippi Plateau Section, f. Mississippi Embayment Section;

(3) Oak-Hickory Forest Region, Southern Division, a. Interior Highlands or Ozark Plateaus, b. Forest Prairie Transition; Northern Division, a. Mississippi Valley Section, b. Prairie Peninsula Suction;

(4) Oak-Chestnut Forest Region, a. Southern Appalachians, b. Northern Blue Ridge, Ridge and Valley Section, d. Piedmont Section, e. Glaciated Section;

(5) Oak-Pine Forest Region, a. Atlantic Slope Section, b. Gulf Slope Section;

(6) Southeastern Evergreen Forest Region, a. Mississippi Alluvial Plain;

(7) Beech-Maple Forest Region;

(8) Maple-Basswood Forest Region, a. Driftless Section, b. Big Woods Section;

(9) Hemlock-White Pine Northern Hardwoods Regions, Great Lakes-St. Lawrence Division, a. Great Lake Section, b. Superior Upland, c. Minnesota Section, d. Laurentian Section; Northern Appalachian Highland Division, e. Allegheny Section, f. Adirondack Section, g. New England Section; B, Boreal Forest.

The forest of regions 1, 2, 3, 4, and 7 are considered the Central Hardwood Forest (Braun 1950).

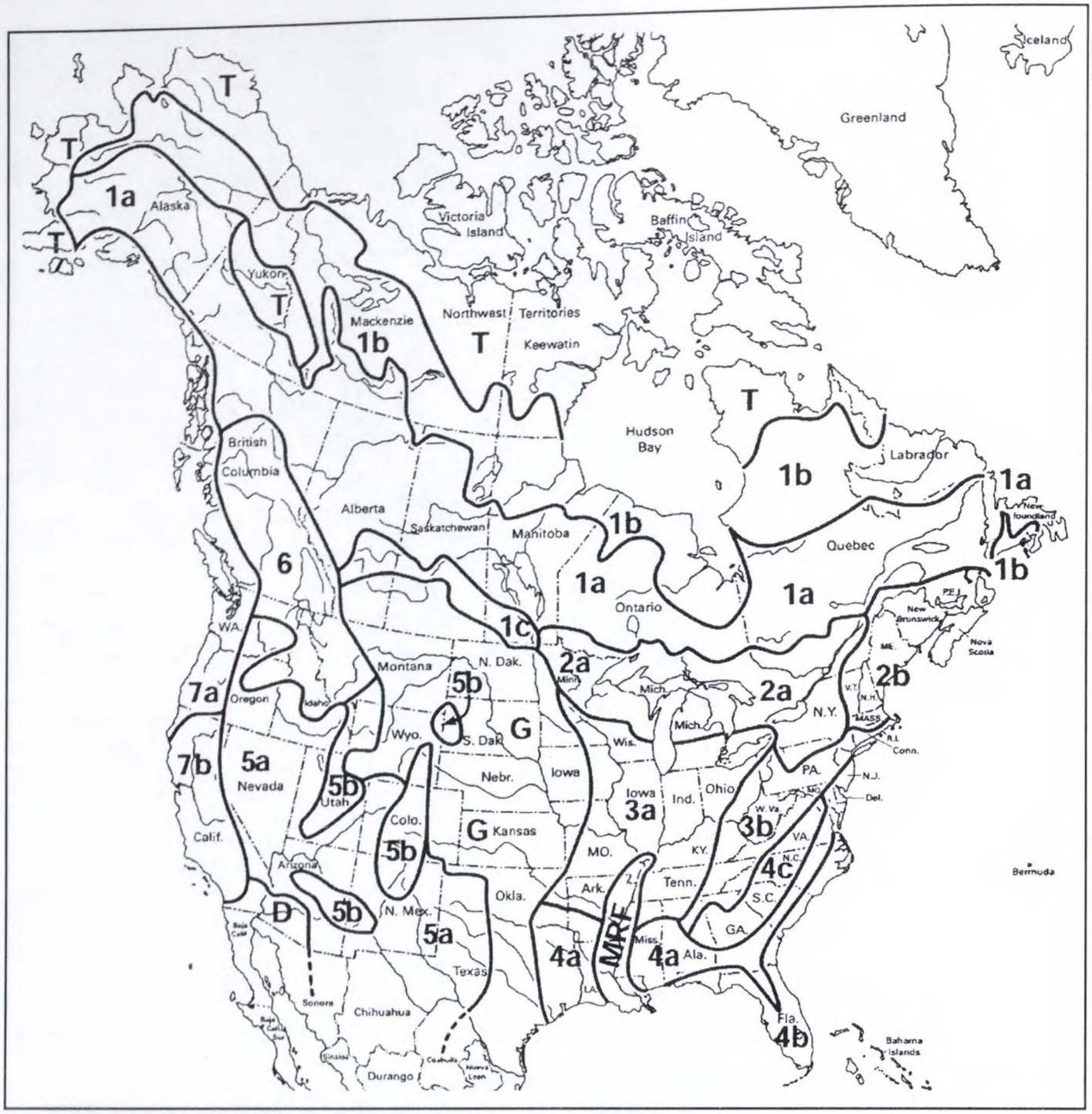

Figure 2.—*Forest regions of North America:*

(1a) Northern Conifer-Hardwood or Boreal (spruce-Fir) Forest, closed; (1b) Northern Conifer-Hardwood forest, open savanna and barrens; (1c) Northern Conifer-Hardwood forest, Aspen Parkland;

(2a) Northern Hardwood-Conifer Forest (Sugar Maple-Yellow Birch-American Beech-White Pine Forest), Great Lakes Section; (2b) Northern Hardwood-Conifer Forest, New England Section;

(3a) Central Hardwood Forest (Oak-Hickory and mesophytic elements), Plateaus section; (3b) Central Hardwood Forest, Appalachian Mountain section;

(4a) Southeastern Pine-Hardwood Forest, Upper Coastal Plain; (4b) Southeastern Pine-Hardwood Forest, Lower Coastal Plain; (4c) Southeastern Pine-Hardwood Forest, Piedmont Plateau;

(5a) Southwestern Juniper-Pinyon Savanna; (5b) Central and Southern Rocky Mountain Mixed Conifer Forest;

(6) Northern Rocky Mountain Conifer Forest;

(7a) Northern Coastal Conifer Forest; (7b).Sierra Nevada Mountain/Southern Coastal Conifer Forest. Other zones include grassland (G), desert (D), and tundra (T), as well as the Mississippi River Valley (MRV) which has been largely cleared for farming (adapted from Fralish and Franklin 2002).

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succession." However, disturbance and successional process do not imply migration of species from one forest region to another. Moreover, there is a definite repeating pattern of climax oak-hickory on thin soils and warm dry south slopes, and a pattern of succession of oak-hickory to maple on deep soils and north slopes; these patterns define most of the region from Missouri to eastern Kentucky (Appalachian Mountains). Climatically, Bailey (1980, 1985, 1994, 1995) places the CHF within the temperate climatic zone of cool to warm winters and warm to hot summers (Humid Temperate Domain, hot continental).

Bailey (1996) states that "Even less satisfactory [than vegetation] is the use of soil types as the basis for a major subdivision." "Soil type frequently does not reflect climate, because the nature of the geologic substratum influences the profile." At the Soil Type level, Bailey is correct; the profiles might change but not the soil developmental processes that operate over a large region. At the Soil Order level, there is a direct correspondence between general soil development, vegetation, and macroclimate. In the oak-hickory dominated CHF, the climate is temperate and the soils generally are classed as Alfisols (gray-brown podzolics) or as Ultisols (red-yellow podzolics) in southern sections (Soil Survey Staff 1975). In the cold Northern Conifer-Hardwood (boreal) forest dominated by spruce-fir-jack pine, the soils are classed as Spodosols (podzols). The Southern Pine-Hardwood Forest soils are Ultisols. The soils of the tall grass prairie are dark Mollisols. The relationship is so strong that regional vegetation and soil conditions could well be used to predict macroclimatic conditions (Albert and Mellilo 1991), or validate the location of ecosystem boundaries.

In terms of physiography, Bailey (1996) indicates "physiographic units cut across energy zones and their associated ecosystems." But there are some excellent examples of physiographic units that functionally identify the boundary of the CHF. Unquestionably, within a particular domain, physiography, associated topography, and soil play a major role in determining subdivisions.

The arguments of Bailey (1996) not withstanding, the most appropriate approach to establishing the location for any forest region is to let "tension zones" define the boundary. Curtis (1959) defined the Wisconsin tension zone as a transition zone (fig. 3) between two regions or vegetational formations (sensu Clements 1928) that contains some components of each region. It is implied that a tension zone is a belt where the ranges for a sufficient number of tree, shrub, herb, and animal species terminate, thus separating two adjoining regions. Animal as well as plant species may be considered in locating the zone. The identified boundary is then delineated by species restricted to each region. Henceforth, the "tension zone" will be referred to as a "transition zone" since it is defined as such.

The importance of a transition zone should not be based on its width or a specific distance, but should vary with the character of the two adjacent forest or vegetation regions. When a zone is a function of a macroclimatic pattern, it may be extremely broad (100 to 200 miles). As noted earlier, vegetation patterns show a strong relationship to macroclimatic patterns, and it can be shown that there is a considerable correspondence between the two sufficient to create a distinct boundary at least for most of the major forested regions.

Conversely, the transition zone is likely to be narrow (5 to 10 miles) when it results from an abrupt boundary that occurs where two contrasting physiographic provinces are in juxtaposition. Changes in geology (bedrock and surface deposition), topography (relief, elevation, and erosion), soil orders and associated types, and drainage/site patterns usually tend to follow and mark boundaries between physiographic provinces. Therefore, for a region the size of the CHF, the boundary may be defined by a combination of transition zones and associated geologic, topographic, edaphic, and atmospheric patterns which do not follow political (state) boundaries but which usually extend across portions of various states.

It is important to emphasize that the large forested regions of North America can not always be defined on the basis of the ranges (presence/absence) of the most important or most common tree species as the ranges of major species may extend into other forested regions. For example, many of the oak (*Quercus* L.) and hickory (*Carya* Nutt.) species of the CHF are important components of the Southern Pine-Hardwood Forest region and are infrequently found in the Northern Hardwood-Conifer Forest. Also, the ranges of sugar maple (*Acer saccharum* Marsh.), basswood (*Tilia americana* L.), American beech (*Fagus grandifolia* Ehrh.), black oak (*Quercus velutina* Lam.), and white oak (*Q. alba* L.) extend into the Northern Hardwood-Conifer, Central Hardwood, and

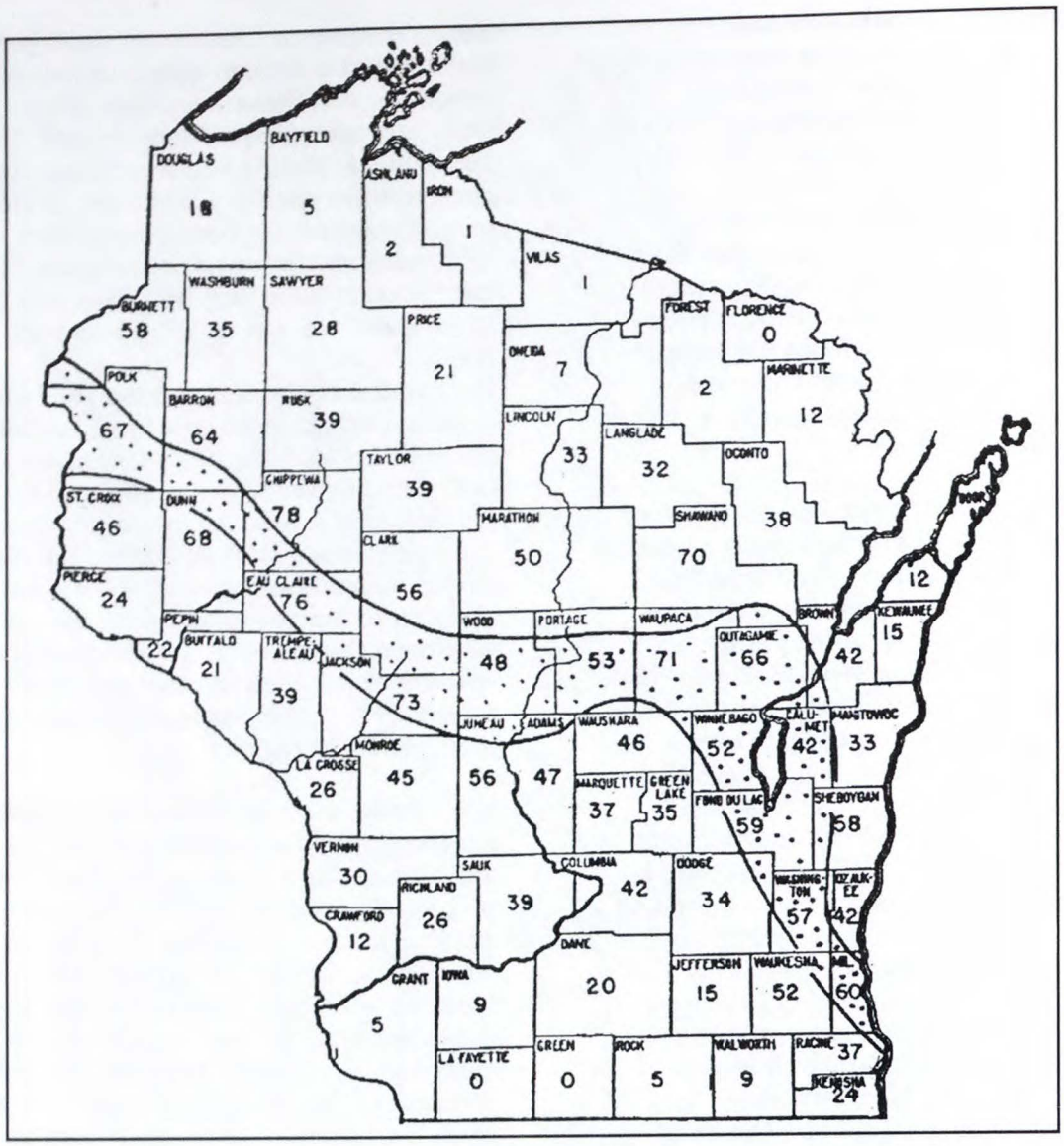

Figure 3.—*The transition (tension) zone between the Northern Hardwood-Conifer Forest and the Central Hardwood Forest in Wisconsin. The number in each county indicates the number of species whose range terminates in that county. Data is for 182 species. The shaded band is the transition zone determined by the greatest number of range terminations (Curtis 1959). Reprinted by permission of the University of Wisconsin Press.*

Southern Pine-Hardwood forests. Often parts of the ranges of major species and species of secondary importance will delineate the transition zone. Moreover, general vegetation as noted by relative importance of species, the pattern of community types, and successional process as well as soil forming processes and soil orders remain relatively constant across a region.

BOUNDARIES OF THE CENTRAL HARDWOOD FOREST

Fralish and Franklin (2002) outlined the CHF region so it distinctly contrasts with those forest and plant communities of adjacent regions, although some parts of the boundary between regions are more distinct than others (fig. 2). The Northern Hardwood-Pine forest of the Lake States (northeast Minnesota, northern Wisconsin, northern Michigan, southern Ontario, northern New York, and most of New England) has been called various names most of which include a reference to pine [white pine; red pine (*Pinus resinosa* Ait.); jack pine (*Pinus banksiana* Lamb)], eastern hemlock or yellow birch (*Betula alleghaniensis* Britton). The region to the south and east, generally including the Gulf and Atlantic Coastal Plain and Piedmont provinces, is called the Southern Pine Region, or perhaps more appropriately, the Southeastern Hardwood-Pine Region as pine has been,

is being, and will continue to be replaced by hardwoods. To the west is the Great Plains, at one time covered with prairie grasses and associated forbs, but now supporting a near monoculture of introduced grasses.

Species restricted to other forest regions not only delineate the boundaries of their respective regions, but also those of the CHF; these regions include the Northern Hardwood-Conifer Forest to the north, and the southern Pine-Hardwood Forest to the east and south. Physiognomically (form and structurally), the CHF is relatively distinct from adjacent regions. Two of the three adjacent regions (Northern Hardwood-Conifer Forest; Southeastern Pine-Hardwood Forest) have five to ten conifer species as a major component. The Tall Grass Prairie Region to the west is, without question, physiognomically distinct although deciduous woodland can be found along streams and drainages (gallery forests). The distinctness of the Tall Grass Prairie-CHF boundary needs no elaboration.

Northern Boundary

In Bailey's (1996) classification, the CHF is placed in the Hot Continental Division of the Humid Temperate Domain. The Northern Hardwood-Conifer Forest directly to the north is in the Warm Continental Division. The line dividing these two forest regions is aligned northwest to southeast from Minneapolis, MN, to about Milwaukee, WI (fig. 3). It is more than coincidence that in Wisconsin, this climatic line follows Curtis' (1959) tension zone separating the area he called the northern floristic province [Northern Hardwood-Conifer Forest of sugar maple, red maple (*Acer rubrum* L.), basswood, yellow birch, and white pine] from the southern floristic province (CHF of white and black oak and hickory). This zone is relatively narrow, varying in width from 10 to 30 miles and midway between the two cities, angles east to west briefly to allow for the inclusion of the old lake bed of Glacial Lake Wisconsin in the CHF. Bailey's climatic line similarly breaks. In part, Curtis documented the climatic change from the northern to the southern floristic provinces when he began to develop northern forest communities at the University of Wisconsin Arboretum (Madison) and found that, while a red and white pine overstory developed, he could not establish the shrubby and herbaceous understory associated with these forests. One suggestion among several for the failure was that rain from the shorter, less frequent storms of the southern province did not penetrate the pine litter and that drought periods were more common such that soil water could not be maintained at the required level for understory plants. Curtis (1959) indicated that the same transition zone could be traced through Minnesota and Michigan. The northern boundaries of black willow (*Salix nigra* Marsh.), eastern cottonwood (*Populus deltoides* Bartr. ex Marsh.), black oak, bitternut hickory (*Carya cordiformis* (Wangenh. (K. Koch), and boxelder (*Acer negundo* L.) closely follow this transition zone eastward into Michigan to Pennsylvania while that of pignut hickory (*C. glabra* (Mill.) Sweet) coincides with the boundary from Michigan through Pennsylvania.

East across Lake Michigan on the western shore of Lower Michigan, the transition zone begins at the about same latitude as that of Milwaukee, and continues in a serpentine fashion east-northeast across Michigan (Veach 1953, 1959). It intercepts the east shore of Lake Huron near Port Huron or about midway between Detroit and the tip of the "thumb" in Huron County. As in Wisconsin, this 30-mile transition zone separates the sugar maple, American beech, basswood, and white pine of the Northern Hardwood-Conifer Forest from the white and black oak and hickory dominated communities of the CHF. Using climatic data, Albert and others (1986) show a distinct boundary about 60 miles north of that shown by Veach (1953, 1959). It would seem that the area between the boundary for the CHF mapped by Veach and the boundary shown by Albert may well be considered the transition zone.

In Ontario, the 30-mile wide transition zone begins on the west shore of Lake Huron near Goderich and Kincardine, Ontario. The narrow CHF area of Ontario between the transition zone and the United States border to the south is only 30 to 70 miles wide. Here the CHF is referred to as the Deciduous Forest region (Rowe 1972, Farrar 1995) or more generally as temperate deciduous forest (Scott 1995). Clayton and others (1977) refers to the area as the West St. Lawrence Lowland physiographic province. The soils of this part of southern Ontario are classed as gray-brown luvisols developed under deciduous or mixed deciduous forest that correspond to the older U.S. class of gray-brown podzolics (Alfisols). Some research manuscripts on this area have been included in the previous CHF Proceedings (von Althen and Webb 1978, 1980; von Althen 1987, 1989).

From its location on the northwest shore of Lake Erie, the transition zone angles south including the western one-fifth of Pennsylvania in the CHF (figs. 1 and 2). From near the border with Virginia (western section), the CHF border angles northeast across New York into Connecticut and Rhode Island to include part of Massachusetts and Cape Cod. This area of the CHF boundary is probably the least well defined because of the elevational, climatic, and soil changes associated with various mountain ranges. While specific location of the transition zone through Pennsylvania and New York varies from map to map, there is nearly universal agreement to include Cape Cod where bear oak (*Quercus ilicifolia* Wangenh.) woodland and pitch pine form communities.

There are a few species restricted to regions outside the CHF whose ranges coincide with th northern part of the CHF boundary as it has been defined here. Species of the Northern Hardwood-Conifer Forest that identify the northern boundary include balsam fir (*Abies balsamea* (L.) Mill), black spruce (*Picea mariana* (Mill) B.S.P.), red pine, and balsam popular (Populus balsamea L). White pine, eastern hem lock and yellow birch are an anomaly in that these species of the Northern Hardwood Forest generally delineate the boundary in Minnesota, Wisconsin, and Michigan, but in the east they become a component the CHF because they extend southward to follow the cool climate of the Appalachian Mountains. Here, the edge of

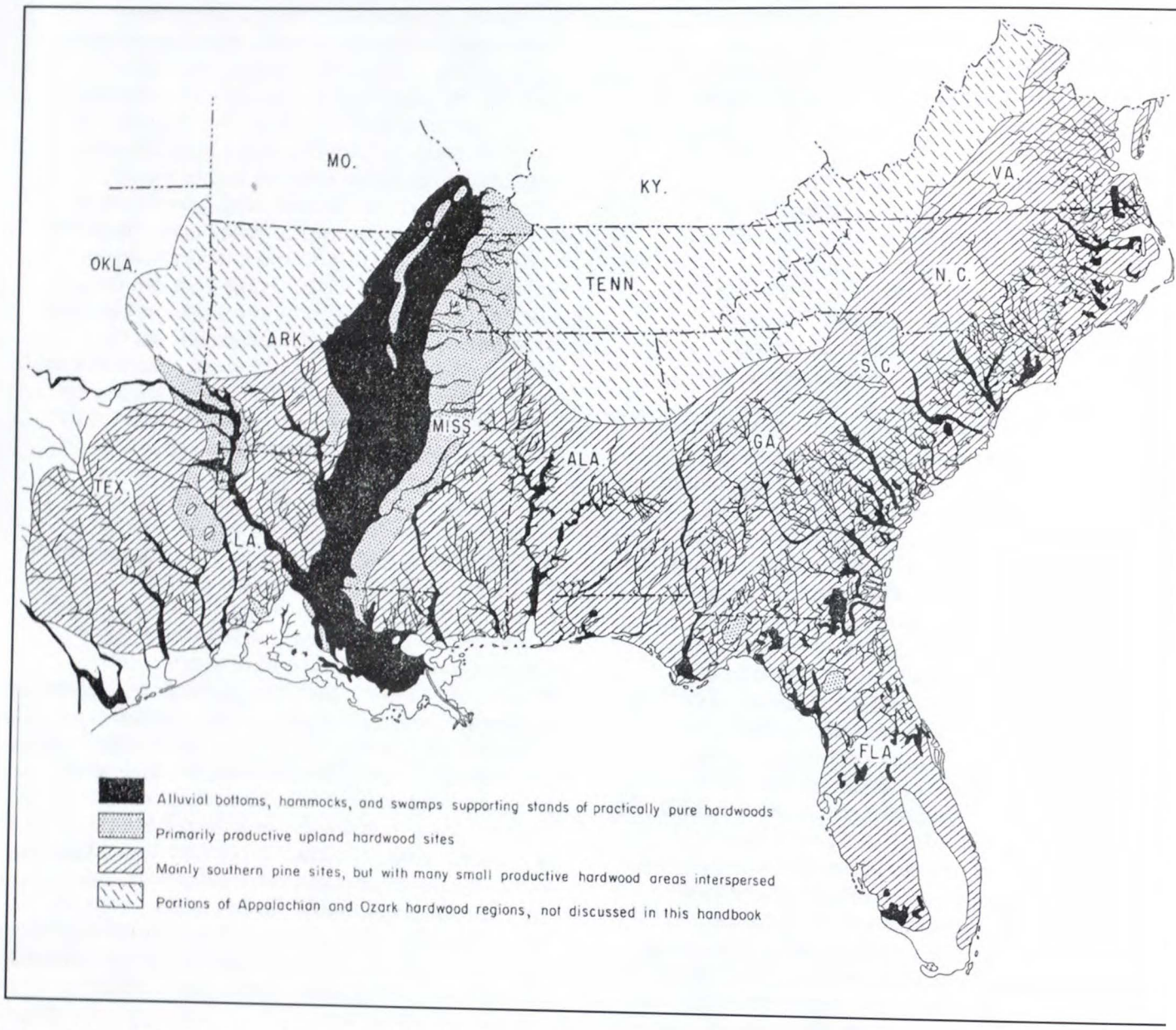

Figure 4.—*Streams and bottomlands of the Atlantic and Gulf Coastal Plain (Putnam and others 1960).*

their range coincides with the boundary between the Blue Ridge and the Piedmont provinces. The northwestern boundary in Michigan, Wisconsin, and Minnesota roughly follows the southern boundaries of white spruce (*Picea glauca* (moench) Voss), jack pine, and northern white-cedar (*Thuja occidentalis* L.), the latter with small scattered outliers in the CHF.

Eastern-Southeastern Boundary

From Long Island, NY, progressing southwest, the transition zone crosses a narrow area of the Atlantic Coastal Plain and then follows the physical boundary between the high rugged Blue Ridge Mountains and the low eroded rolling Piedmont Physiographic province (figs. 1, 2, and 4). The boundary and its associated provinces extend southwest into northern Georgia as delineated by Hicks (1998). This eastern-southeastern boundary has not only a physical/topographic distinction and separates the CHF from the Southeastern Pine-Hardwood Forest (Fralish and Franklin 2002), it also separates Entisols and Alfisols of the Blue Ridge province from the Ultisols of the Piedmont province. The Blue Ridge Mountains are vegetatively distinct from the Piedmont province because of the concentration of loblolly pine (*Pinus taeda* L.) and shortleaf pine forest mixed with oak in the latter. Both Braun (1950) and Waggoner (1975) indicate that the Piedmont coincides with the eastern oak-pine transition zone, which continues through Georgia and Alabama into Mississippi.

Southern Boundary

In northern Georgia, the boundary angles westward to encompass the southern extensions of the high Blue Ridge Mountains, Ridge and Valley, and Appalachian Plateau in north-central Alabama (figs. 1 and 2). From here, the boundary traditionally has been angled northwest to intersect the Brown Loam Hills region in northwest Mississippi. The actual transition zone of 50 percent pine and 50 percent oak-hickory across the northern sections of Georgia, Alabama, and Mississippi is relatively broad (80-120 miles); this transition zone is sufficiently large that it has been referred to as the "oak-pine region" (Braun 1950, Waggoner 1975). As delineated here, this area is considered outside the CHF although a strong argument to the contrary can be made since the headwaters of the major streams are generally located near or in the oak-pine transition zone (fig. 4). Concurrently, these headwaters also mark the northern extent of various southern bottomland species found on the terraces of these streams in the Gulf Coastal Plain; thus in addition to the oak-pine zone, there is a change in bottomland forest composition and physiographic provinces across the forested landscape that delimits the CHF southern boundary. The exception to this pattern is the Mississippi embayment (floodplain) that extends like a thumb north into southern Illinois and includes a narrow strip of west Kentucky and west Tennessee. A few southern bottomland species follow this floodplain north into the CHF.

In Georgia, Alabama, Mississippi, and Arkansas, the southern CHF boundary is also the northern boundaries of loblolly pine, blue-jack oak (*Quercus incana* W. Bartrom), and laurel oak (*Q. hemisphaerica* Bartr.) of the upland Southeastern Pine-Hardwood Forest. Southern bottomland species that follow the CHF boundary include sweetbay (*Magnolia virginiana* L.), overcup oak (*Quercus lyrata* Walt.), swamp chestnut oak (*Q. michauxii* Nutt.), cherrybark oak (*Q. pogoda* Ell.), and water oak (*Q. nigra* L.).

An extension of the east-west boundary across the Brown Loam Hills and the Mississippi floodplain (both exclusively hardwood regions) into central Arkansas and Oklahoma essentially completes the southern boundary. The traditional dividing line between the CHF and the Southeastern Pine-Hardwood Forest is the Arkansas River Valley between the Boston Mountains to the south, and Ouachita Mountains to the north. The forest of the Boston Mountains of central Arkansas is oak-hickory with a mixture of southern pine while that of the Ouachita Mountains is mapped as pine-oak transition (Shantz and Zon 1924). Bruner (1931) as cited in Braun (1950) states that the pine stands of this area "represent the northwestern extension of the southern pine forest."

Western Transition Zone

The Western Transition Zone is formed from a mixture of two physionomically different plant communities: prairie (grassland) and oak-hickory savanna or woodland (trees). The cross-timbers area extends from northeast Texas east of the panhandle into a wide area covering much of Oklahoma, and terminates in a narrow section of southeast Kansas (Hoagland and others 1999). At its maximum width in Oklahoma, the Cross Timbers region is over 250 miles wide including disjunct areas. The primary species of the Cross Timbers are post oak (*Q. stellata* Wangenh.) and blackjack oak (*Q. marilandica* Muenchh.) with black oak, and black hickory (*Carya texana* (Le Conte) DC) of

secondary importance. However, a broad expanse of intermixed oak-hickory woodland (savanna) and prairie (the prairie-forest border region) continues northward along an area west of Missouri and Iowa into Kansas and Nebraska. Here most forested areas are along stream drainages (gallery forest). The ranges of a large number of species extend into the Western Transition Zone (Little 1971). The western range of bur oak (*Q. macrocarpa* Michx.) is most closely aligned with this zone from Texas to Minnesota (fig. 5).

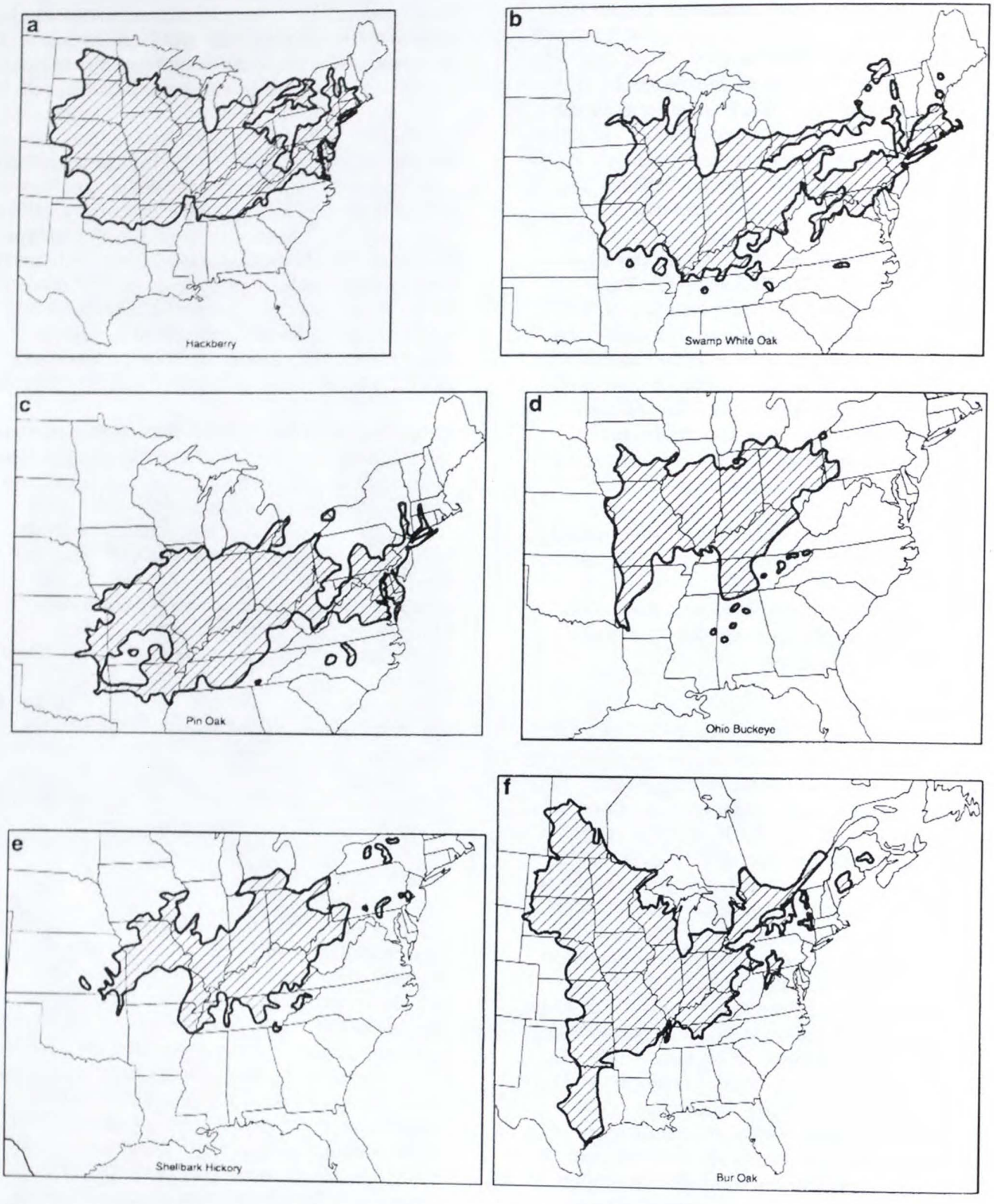

Figure 5.—*Hardwood species whose ranges are restricted to and collectively delineate the Central Hardwood Region (Little 1971).*

In northern Missouri, the zone broadens and extends eastward and across a large area of south-central, central, and northern Illinois. Early records indicate this transition zone, the "prairie-forest border region," extended through much of southern Wisconsin into north-central Indiana (Anderson 1982). Disjunct areas also once existed in southern Michigan, Ohio, and Kentucky. This zone also has been called the "prairie peninsula" (Transeau 1935). The zone continues as a narrow strip of oak-hickory savanna intermixed with maple-basswood forest that extends from the tri-state area (northwest Illinois, northeast Iowa, and southwest Wisconsin) into Minnesota (Kuchler 1964, Bailey 1976, 1994).

COMMONALITY OF COMMUNITY COMPOSITION AND SUCCESSIONAL PROCESSES

One general characteristic that connects most parts of the region outlined above (fig. 2) is the extensive distribution of high quality (i.e., site index 50-80) oak-hickory communities (stands) on xeric-mesic (intermediate dry-moist) and mesic (moist) sites (deep soil, north, northeast, and east slopes, low slopes, drainages, stream terraces). Nowhere else does a concentration of this magnitude occur. The Blue Ridge Mountains with their conifer component of hemlock and white pine in coves and Fraser fir (Abies fraseri (Pursh) Poir) at high elevations are integrated into the CHF due to the surrounding lower elevation oak-hickory zones. Similarly, oak-hickory dominated areas surround other concentrations of conifers such as pitch pine and Virginia pine. Also areas of poor quality (site index 35-50) oak-hickory stands on xeric (droughty) sites characterized by thin rocky soils, exposed south, southwest, and west slopes, and/or locations in the prairie-forest transition zone, but the strong domination by oak integrates these into the CHF region. The same argument applies to the bottomland forests that have a strong oak component.

For those unfamiliar with the CHF and its forests, the 12 major upland oak species include white, post, chestnut (*Q. prinus* L.), chinkapin (*Q. muehlenbergii* Engelm) in the "white oak" group, and in the "red oak" group, black, northern red (*Q. rubra* L.), southern red (*Q. falcata* Michx.), scarlet (*Q. coccinea* Muenchh), Hill's (*Q. ellipsoidalis* E.J. Hill), shingle (*Q. imbricaria* Michx.), blackjack, and bear oak. In the bottomland forests, there are six common species: swamp white (*Q. bicolor* Willd.), swamp chestnut, overcup, pin (*Q. palustris* Muenchh), willow (*Q. phellos* L.), and Shumard (*Q. shumardii* Buckl.), the first three are in the "white oak" group and the latter three in the "red oak" group. The 10 hickory species include shagbark (*C. ovata* (Mill) K. Koch), pignut, red (*C. ovalis* (Wangenh) Sarg.), bitternut, mockernut (*C. tomentosa* Nutt), sand (*C. pallida* (Mill) Engl.), and black hickory (*C. texana* (Le Conte) DC) in upland forest and water (*C. aquatica* Nutt), pecan (*C. illinoiensis*) Wangenh.) K. Koch, and shellbark hickory (*C. laciniosa* (Michx. F) Nutt) in bottomland forest. Stems of shagbark, shellbark, and mockernut hickory are occasionally found on both upland and bottomland sites.

In addition to composition, community succession also is a process unifying a forest region. In the Northern Conifer-Hardwood Forest (boreal forest, fig. 1) that is transcontinental across Canada, the species of the compositionally stable (climax) forest are white spruce (*Picea glauca* Michx.) and balsam fir. But after disturbance by fire or cutting and fire, the upland forest may be composed of trembling aspen (*Populus tremuloides*), black spruce, or jack pine. This replacement is a unifying process in the boreal forest.

In the Lake States Northern Hardwood-Conifer Forest, sugar maple, basswood, hemlock, white pine, and American beech in eastern Wisconsin and Michigan are dominant species on medium and fine textured soils. Although the severe timber harvesting and fire of the late 1800s and early 1900s cleared this forest, it was quickly replaced by trembling aspen and bigtooth aspen (*Populus grandidentata* L.), and to a lesser extent, white birch (*Betula papyrifera*) after fire control laws were enacted in the 1920s and 1930s. However, at present the northern hardwoods are rapidly replacing aspen and birch as the dominant community type unless silvicultural treatments reverse the process to maintain these shade intolerant species. On coarse sands, a mixture of white pine, northern red oak, and red maple appear destined to form compositionally stable stands, often replacing relatively pure stands of aspen, jack pine, or Hill's oak. These successional processes tend to unify the Northern Hardwood-Conifer Forest.

In the Southeastern Pine-Hardwood Forest, the major pine species are shade intolerant or extremely shade intolerant, and thus, are early-successional species that are easily and rapidly being replaced by a variety of oak and other shrubby species (e.g. holly (*Ilex* spp.). These species need a high level of disturbance for seedlings to become established and grow into mature overstory trees.

In the CHF, oak and hickory species have an unusual ecological role in that they form both compositionally stable and successional stands. Neither the aspens of the Northern Hardwood-Pine region nor the pines of the Southeastern Pine-Hardwood Forest are found in both conditions. Oak and hickory form compositionally stable (climax) woodlands on xeric sites and xeric-mesic sites while stands composed of many of the same species are successional on mesic sites. Thus, in the region outlined, oak is an ever-present unifying component over the landscape. Braun (1950) and Bryant and others (1993) describe the forest communities associated with the physiographic provinces described later.

The present oak forests on mesic sites apparently developed after a long period of continuous disturbance by fire and grazing, and later by timber harvesting. As with southern pine and aspen, the various oak species are shade intolerant or extremely intolerant and are early to mid-successional species. Seedlings require moderate light levels to become established, but as they grow they become more intolerant, thus high light levels must be maintained until oak-hickory stems reach the overstory canopy. In the western half of the CHF, forests composed of mesophytic species such as sugar maple, American beech, bitternut hickory, and red elm (*Ulmus rubra* Muhl.) form a dense shade and are rapidly replacing oak forest on mesic sites. In the eastern half of the region, red maple and white pine tend to be the late successional species. On xeric sites, oak and hickory often continue to dominate because of the lack of competition.

There is a seemingly paradoxical relationship between oak growth and its replacement pattern. Growth is slowest and site index lowest (35-50) on xeric sites and intermediate (site index 50-65) on xeric-mesic sites. Growth is fastest and site index highest (65-80) on mesic sites. Therefore, it would appear logical that with the rapid growth of oak on mesic sites, it would be relatively easy to manage for these species since they would outgrow nearly all other species. The problem is that the moist site conditions are also ideal for establishment of a dense understory canopy of mesophytic species that usually form a dense shade and prevent the establishment of oak seedlings. In the CHF, the most productive oak stands are being replaced by mesophytes, and this process is of major concern. Conversely, since mesophytic species do not survive on xeric and xeric-mesic sites, the oak stands on these sites remain relatively compositionally stable.

There is an interesting linkage between shade tolerance and drought tolerance (Fralish 1988). Shade intolerant (light demanding) species are usually drought tolerant and able to withstand stress associated with depleted soil water levels. Because of the severe environmental conditions, mesophytic species can not invade, forest density remains low, light levels at the forest floor remain high, and oak seedlings may grow and develop. Most mesophytic (moisture demanding) species are shade tolerant because on moist, deep soil sites, they must be able to survive at extremely low light levels in a dense forest composed of many species. Physiologically, the survival mechanisms between the two groups of species are quite different.

SPECIES RESTRICTED TO THE REGION

Although many of the oak and hickory species listed above have ranges that extend well beyond the CHF, there are a variety of both upland and bottomland species that are completely or largely restricted to within the delineated boundaries (fig. 5). No one single species has a range that perfectly matches the CHF boundaries. However, the ranges of a number of species are close to matching the boundaries west of the Appalachian Mountains and collectively their ranges can be used to delineate nearly all of the CHF. This list includes hackberry (*Celtis occidentalis* L.), bur oak, chinkapin oak, swamp white oak, shellbark hickory, and Ohio buckeye (*Aesculus glabra*). Individually, the ranges of hackberry, chinkapin oak, and bur oak come the closest to matching the CHF boundary but the range of hackberry extends into Vermont, New Hampshire, and northern New York, bur oak into the Northern Hardwood-Conifer Forest of southern Canada from Manitoba to Quebec, and chinkapin oak into the Southern Pine-Hardwood Forest from South Carolina to Mississippi.

Species whose eastern range boundary follows the interface between the Blue Ridge Mountains and the Piedmont province include yellow buckeye (*Aesculus flava* Ait.), Fraser magnolia (*Magnolia fraseri* Walt.), and black locust (*Robinia pseudoacacia* L.).

The ranges of black maple (*A. nigrum* Michx.), dwarf chinkapin (*Q. prinoides* Willd.), and butternut (*Juglans cinerea* L.) also are closely aligned with the CHF boundary except these species extend into New York, Ontario, Quebec,

and parts of New England. Scarlet oak and chestnut oak are CHF species including the Appalachians except that their distribution also includes the Piedmont. Most of the range of Hill's oak in Wisconsin and Minnesota is within the CHF. Shagbark hickory is primarily a CHF species although it can be found in New England and Quebec, on the Piedmont, and in sections of the Southern Pine-Hardwood Forest. The extension of the ranges of shagbark hickory, black maple, and northern red oak into southern Quebec may have been a factor in Leopold and others (1998) considering it as part of the CHF.

Other species with relatively small or restricted distributions inside the CHF that do not coincide with the boundary include northern catalpa (*Catalpa speciosa* Warder), blue ash (*Fraxinus quadrangulata* Michx), Kentucky coffeetree (*Gymnocladus dioica* (L.) K. Koch), pin oak, shingle oak, bear oak, and yellowwood (*Cladrastris kentuckea*).

Midcanopy species of the CHF include pawpaw (*Asimina triloba* (L.) Dunal), roughleaf dogwood (*Cornus drummondii* C.A. Meyer), eastern wahoo (*Euonymus atropurpureus* Jacq.), sweet crabapple (*Malus coronaria*), Iowa crabapple (*M. ioensis*

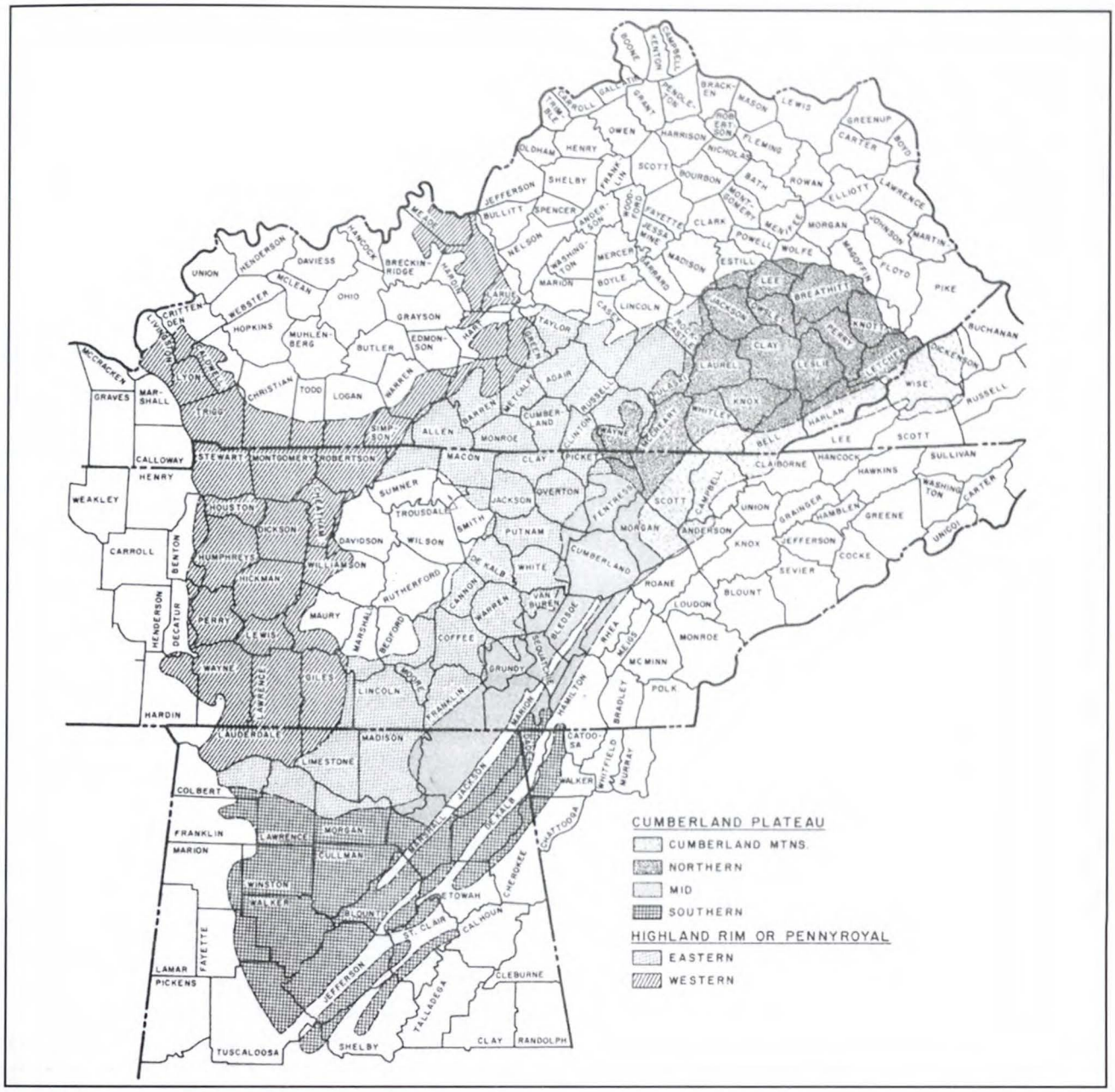

Figure 6.—*Location of the Cumberland Mountain and Plateau and Interior Low Plateau physiographic provinces (Smalley 1979).*

Marsh), American plum (*Prunus americana* Bailey), hortulan plum (*P. hortulana* Bailey), American bladdernut (*Staphylea trifolia*), and common prickly-ash (*Zanthoxylum americanum* Mill.). A number of rhododendron and azalea species and other endemics are found only in the Appalachian Mountains of the CHF.

PHYSIOGRAPHY OF THE CENTRAL HARDWOOD FOREST

Blue Ridge Mountains

The CHF boundaries described above enclose a diversity of bedrock types and soil orders. Beginning at the east boundary, the Blue Ridge physiographic region (fig. 1) extends from southeastern Pennsylvania to northeastern Georgia and has a width of from 5 miles (Northern Virginia) to about 70 miles near Asheville, North Carolina (Buol 1973). The Blue Ridge Mountain range is the eastern-most range of the Appalachian highlands and has the highest peaks. The province consists of various small ranges that are the oldest in the United States. Most peaks are worn to rounded tops and domes, thus, these mountains are referred to as "subdued" (Fenneman 1938). Probably the best known of the mountain complex is the Great Smoky Mountains (National Park) in Tennessee and North Carolina.

Rock types include Precambrian granite and gneiss, siltstone, sandstone, and conglomerate. Most of these rocks have weathered into acidic soils but a few rocks such as marble have weathered into basic soils. The soils are classed either as Ultisols (low base and high clay accumulation in the B horizons) or as Inceptisols (poorly developed profiles).

Ridge and Valley Province

The Ridge and Valley physiographic province (fig. 1) parallels and lies west of the high Blue Ridge Mountains; it parallels and lies east of the eroded Appalachian Plateau. The area is world famous for its fold mountains. As the name indicates, the province consists of a group of long, narrow, usually wooded steep mountain ridges separated by level valley floors. The resistant sandstone bedrock forms the ridges and the weaker shale and limestone the valley floors, the latter eroded to a common level (Buol 1973). In Tennessee, the broad valleys and narrow ridges of the eastern section are called the "Great Valley." The soils of the Ridge and Valley are classed as Ultisols and Inceptisols with some Alfisols in more northern regions.

Appalachian or Cumberland Plateau

Directly west of the Ridge and Valley is the Appalachian Plateau (fig. 1). Along its western boundary are the Interior Low Plateau and the Eastern Central Lowlands. The area has been divided into the Southern Cumberland Plateau (Smalley 1979), Mid-Cumberland Plateau (Smalley 1982), Cumberland Mountains (Smalley 1984), and Northern Cumberland Plateau (Smalley 1986). The area is underlain in many places by horizontal sandstone and the highly dissected topography is characterized by dendritic drainage with winding narrow-topped ridges and deep narrow valleys. In some areas the sandstone bedrock creates upland flats (Buol 1973). Collectively, the four subregions are oriented southwest to northeast and parallel the Ridge and Valley province. The soils of the Plateaus are primarily Ultisols (Hapludults) with Inceptisols (Dystrochrepts) found in the eastern area. The soils of the Cumberland Mountains are primarily Inceptisols (Dystrochrepts) and Ultisols (Hapludults), along with rockland (Buol 1973).

Interior Low Plateaus

The Interior Low Plateaus physiographic province has been intensively studied by Smalley (1980, 1983), DeSelm and Schmalzer (1982), and Quarterman and Powell (1978). The Appalachian Plateau bound the province to the east and south (fig. 6). The western boundary in Kentucky and Tennessee follows the Tennessee River and interfaces with the Coastal Plain sediments of the Mississippi Embayment. The northern boundary is considered to generally follow the southern limit of glaciation through southern sections of Ohio, Indiana, and Illinois (Hunt 1978).

Within the province, a number of different regions are delineated. In the southern part of the triangle are the Eastern Highland Rim and Western Highland Rim enclosing the Central (Nashville) Basin. The northern part of the triangle encompasses the Inner Bluegrass and Outer Bluegrass regions of north central Kentucky. The Western Coal Field along with the Shawnee Hills section and parts of the Western Highland Rim and Pennyroyal form the west section of the triangle that extends into southern Illinois. The Pennyroyal is situated between the Western Highland Rim and the Western Coal Fields. Quarterman and Powell (1978) describe additional subdivisions for each of these regions. Both the Bluegrass subregion and the Nashville Basin are underlain by Ordovician limestone that is often exposed at

the surface. Because of weathering, caves are common and the land surface is pitted with sinkholes (Karst topography). The soils of the Interior Low Plateaus are classed as Alfisols (primarily Paleudalfs) and Ultisols (primarily Hapludults).

Central Lowland Physiographic Province

The Central Lowlands was delineated by Fenneman (1938) and includes the northern midwest from North Dakota east into northeast Ohio and New York and south into central Texas. Vegetatively, this province includes the beech-maple, oak-hickory, Maple-Basswood regions (Braun 1950) and the zone known as the prairie-forest transition (Anderson 1982). The vegetation of the zone west of the prairie-forest border is mid-grass prairie and outside the area of CHF. Although not reviewed here, the Central Lowlands is subdivided into a variety of natural divisions (Lindsey 1966, Schwegman 1973, Homoya and others 1985, Albert and others 1986, Nelson 1987, Bailey 1994, Hole and Germain 1994, Albert 1995). Most states have developed a system of natural divisions.

East Central Lowlands

In the United States, the boundaries of the Eastern Central Lowland Physiographic province are based on those established by Fenneman (1938). Some sections of the boundary are physiographically well defined while others are more or less arbitrary. For convenience, the Mississippi River has been defined as the western boundary. The surface of the Eastern Central Lowlands was predominately formed by erosional and depositional processes, primarily glaciation combined with surface water movement. The glaciers scoured lake basins and lowlands but deposited material that created various formations such as till plains, end and ground moraines, outwash, eskers, drumlins, and kames. These deposits substantially reduced the importance of underlying bedrock.

Older glacial deposits cover most of northern, southern, and western Illinois; northern, central, southwestern, and southeastern Indiana; and the northern and western half of Ohio (Harris and others 1982). Many of these deposits are covered by Wisconsinan age material. The Wisconsinan age ice covered the northeastern half of Illinois (Schwegman 1973) and the northern two-thirds of Indiana (Lindsey 1966) thus leaving older deposits to the south undisturbed. In Ohio, material from the Wisconsinan glacier covered nearly all of that deposited by older glaciers. Capping the surface of the Driftless Area and glacial deposits is loess, a silt or silt loam material blown out of the floodplains of the major river valleys such as the Mississippi, Ohio, and Wabash Rivers. Because of the westerly wind, the deepest loess deposits are found directly east of the rivers and become progressively thinner with distance from the floodplain. In large areas of the East Central Lowlands, the soils are classed as Alfisols but in western, south central, north central, and northern Illinois and in the southeastern Wisconsin area, the soils are a mixture of Alfisols and Mollisols (prairie-forest transition), and Entisols in the lake bed of the old glacial Lake Wisconsin (Hole 1976, Fehrenbacher and others 1984).

The "Driftless Area" of southwestern Wisconsin (fig. 7), northwestern Illinois and northeastern Iowa (West Central Lowland) is the only unglaciated part of the Eastern Central Lowland. Apparently the deep troughs of Lake Superior and Lake Michigan directed the glaciers to the west and east, respectively. One other interesting area in the Eastern Central Lowlands is the lake bed of extinct glacial Lake Wisconsin. During the last glacial advance, the Wisconsin River in central Wisconsin was flooded and backed up resulting in a deposit of deep sand from glacial melt water.

Western Central Lowlands

Near the end of the Mesozoic era, erosion from the Rocky Mountains resulted in the deposition of material that formed a huge east-sloping plain that stretched to the Mississippi River (Risser 1981). The second major influence on this region was glaciation. There is a gradient of vegetation change from east to west with the concentration of hardwood forest in the prairie-forest transition zone of eastern Missouri, Iowa, and central Minnesota decreasing to the west and becoming essentially non-existent except for gallery forests in eastern Kansas and Nebraska. Concurrently, there was an increase in prairie vegetation from east to west; most of the prairie land has been plowed for agricultural crops. The proportion of forest and prairie soils, Alfisols, and Mollisols, respectively, follow the same pattern.

Interior Highlands

The Interior Highlands consists of two unglaciated regions: the Ozark Plateau and the Ouachita Mountains (fig. 8). As noted earlier, the forest of the Ouachita Mountains is a mixture of southern pine and oak-hickory; it is part of the transition forest between the Southeastern

Pine-Hardwood region and the CHF. The Ozark Plateau (often called the Ozark Hills) covers the northwest half of Arkansas, southern half of Missouri, and an eastern section of Oklahoma. The region consists of rugged hills, steep slopes, and deep valleys (Hunt 1974). Along the southern edge of the Plateau are a series of hills widely known as the Boston Mountains. In Missouri, there are a number of subregions (Nelson 1987). The Ozark Plateaus are underlain by several formations of limestone and dolomite of Ordovician, Silurian, Devonian, Mississippian, and Pennsylvanian age; these strata vary with the subregion. The soils are classed as Ultisols, more specifically, Hapludults, Fragiudults, and Paleudults. Soils of steep slopes are Incep-tisols, primarily Dystrochepts (Buol 1973). Some Mollisols (prairie soils) are interspersed.

OTHER RELATED HARDWOOD AREAS

Mississippi Embayment

The Mississippi Embayment (Gulf Coastal Plain) extends from the Delta of the Mississippi River

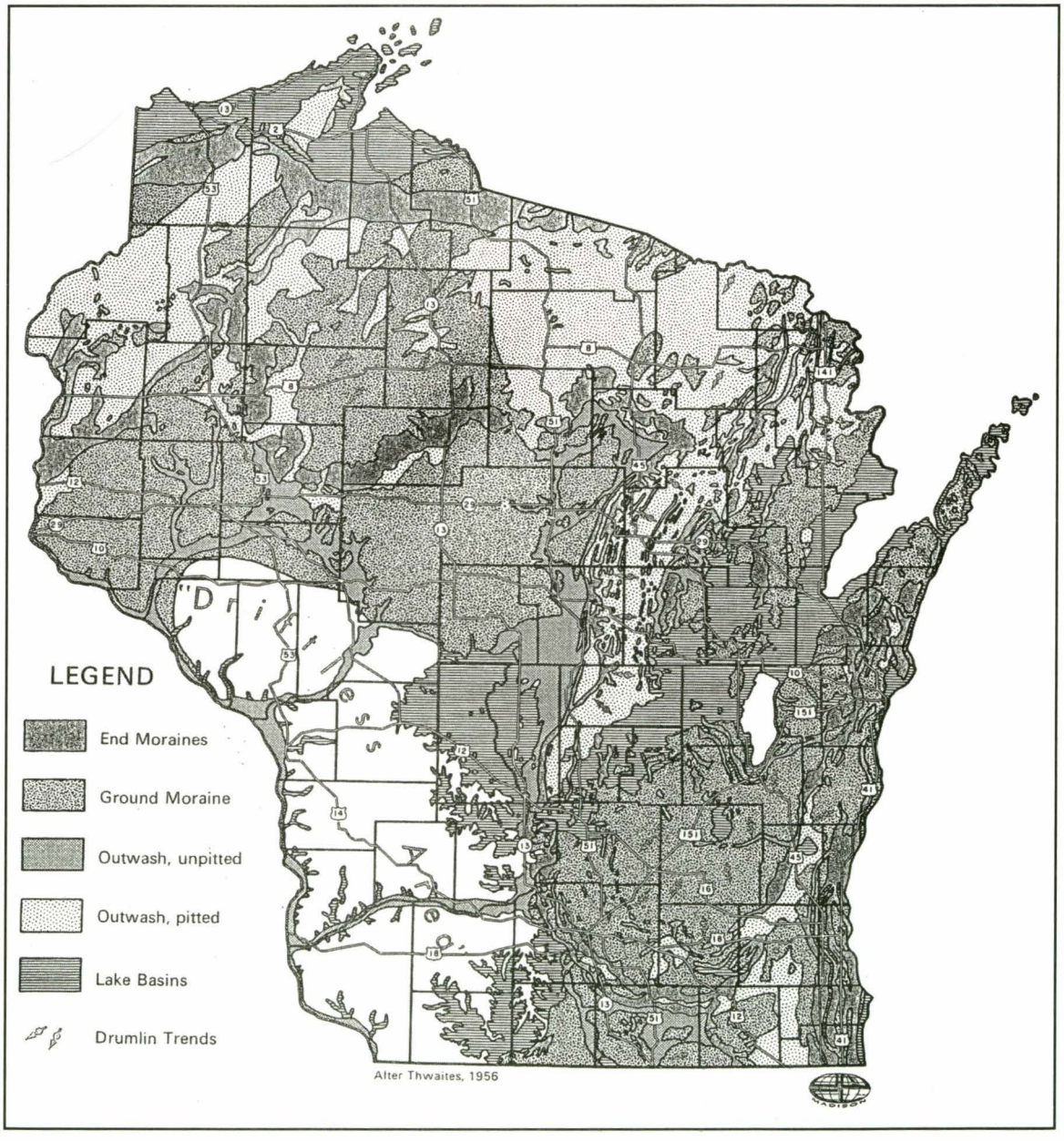

Figure 7.—*Glacial deposits and the Driftless Area of Wisconsin. The lake basin of ancient Lake Wisconsin in the center of the state was created when the glacier blocked the Wisconsin River (Hole 1976). Reprinted with permission of the University of Wisconsin Press.*

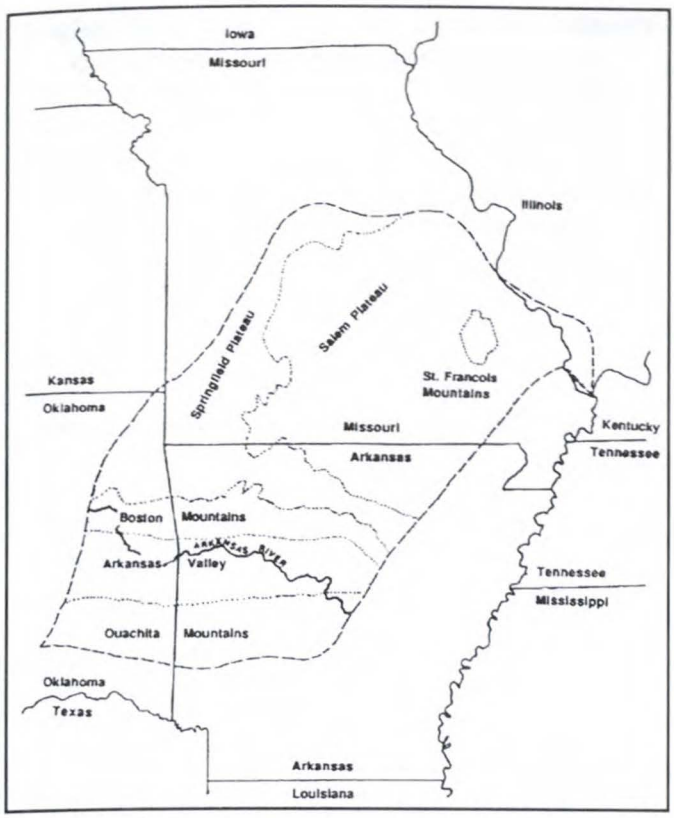

Figure 8.—*Subdivisions of the Interior Highland physiographic province (Zachry and others 1979).*

northward along the floodplain into southern Missouri, Illinois, and Indiana (figs. 4 and 5). A number of bottomland species of the Coastal Plain region grow north of the Arkansas River, the southern boundary of the CHF. This list of species includes willow oak, water oak, cherrybark oak, Nuttall oak (*Quercus nuttallii* Palmer), swamp chestnut oak, overcup oak, swamp tupelo (*Nyssa aquatica* L.), and baldcypress (*Taxodium distichum* (L.) Rich.). The soils are classed as Inceptisols, primarily Haplaquepts. In presettlement time, most of this area was covered by bottomland forest, but because of the deep soil and high fertility most of the forest was cleared for agriculture. One major feature of the floodplain is Crowley's Ridge that stretches south from the Missouri border into central Arkansas (Fenneman 1938, Waggoner 1975).

Loess Bluffs

The Mississippi alluvial plain is bordered on the east by Loess Hills (also called Brown Loam, or Loessial Hills) that are 5 to 15 miles wide extending from Mississippi (Tunica Hills) to Kentucky (Cane Hills) (figs. 4 and 5). They rise 125 to 250 ft above the floodplain and are dissected with ravines particularly along the western margin. The mantle of loess is up 100 ft thick in places. The silt loam material of the bluffs blew out of the floodplain during dry periods, usually during the winter months. Loess thickness on the west bluffs is thinner because the prevailing wind direction is west to east. These deep, fertile, moist soils are classed as Alfisols (Fragiudalfs). They are yellowish-brown in color and contrast with the nearby red-yellow Ultisols. The forest is strongly dominated by mesophytic hardwoods (Braun 1950, Waggoner 1975).

SUMMARY

Although the location of the boundary for the CHF is going to be interpreted somewhat differently depending on the objectives of independent researchers, the region is ecologically distinct and there is considerable commonality in forest community types, successional patterns, and to a lesser extent, physiographic provinces and soil. Although done at different times and working with different base factors (climatic, vegetative, physiographic), there is considerable correspondence among the maps of Braun (1950), Kuchler (1964), Keys and Carpenter (1995), Bailey (1976, 1994), and Fralish and Franklin (2002). The eastern boundary of the CHF reported by Fralish (1994) does not coincide with these maps because only the region of CHF bottomlands was considered thus leaving out the Appalachian provinces.

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SHIFTS AND FUTURE TRENDS IN THE FOREST RESOURCES OF THE CENTRAL HARDWOOD REGION

Thomas L. Schmidt and William H. McWilliams[1]

ABSTRACT.—Forests in the Central Hardwood region are undergoing change in terms of area, volume, species composition, and forest structure. These forests are dominated by deciduous species; are increasing their average stand size, volume, and age; and, are experiencing woody plant species replacement as shade intolerant species are being replaced by more shade tolerant species. As changes progress, concerns are being raised regarding the potential for these lands to produce the wide array of benefits associated with timberland. Changes in the forest resource as well as changes in land ownership and management objectives have repercussions for wildlife species, timber industries, recreational interests, and the wide array of other benefits reaped from this invaluable resource.

The Central Hardwood region, well known for its high quality hardwoods, has a wide variety of forest resources that make significant environmental and economic contributions. These forest resources play an important role with impacts ranging from employment and other value-added economic contributions to improving and protecting soil and water resources to providing wildlife habitat.

Central Hardwood forests are constantly undergoing change due to both natural processes and human impacts. As these changes progress, concerns related to forest fragmentation, species composition, ownership and resulting management objectives, and future sustainability are being raised. The area of forest land throughout the Central Hardwood region is increasing. Total area of forest land in this region steadily decreased from the time of European settlement until the late 1970s and early 1980s primarily due to conversion to agriculture. Since then, the area of forest land has been increasing at a steady rate as reversions from agriculture outweigh forest loss. However, sites where expansion is occurring are different from sites where forests have been lost. As a consequence, the potential for these sites to produce the wide array of benefits associated with forest land is different from what has historically been produced.

With the control of wildfires and selective harvesting methods, these forests are undergoing natural succession and maturing. Successional processes are moving the forest species composition from a mix of early successional species to domination by more shade-tolerant species. Changes in land ownership, management patterns, and harvesting techniques have crucial impacts on the composition and structure of Central Hardwood forests. Because changes dictate the quality and quantity of the future resource, it is important to document these changes.

METHODS

The Central Hardwood study region in this report is defined as the states of Illinois, Indiana, Iowa, Missouri, and Ohio (fig. 1). This region encompasses most of the mid-continental region of the USA. Although other states fall in the Central Hardwood region, a lack of data and the authors' sense that trends occurring in the above states represent trends across the region, and so additional states are not included in this study. Data for this study are from the USDA

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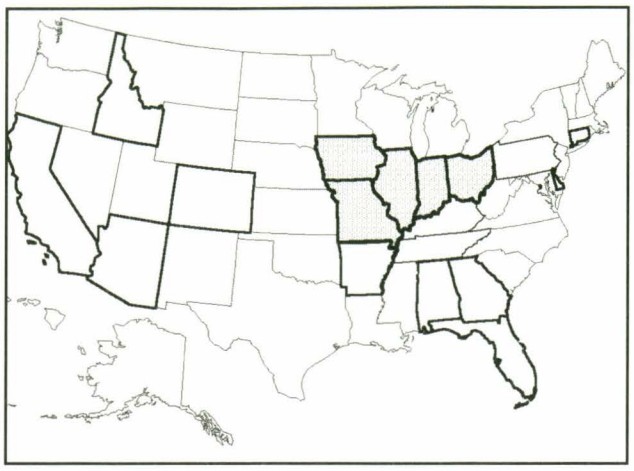

Figure 1.—*Central Hardwood five-state study area.*

Forest Service, Forest Inventory and Analysis (FIA) Database (Miles and others 2001), and are based on the previous and most recent complete statewide forest inventories for these five states. The dates of the FIA inventories are Illinois 1985 and 1998, Indiana 1986 and 1999, Iowa 1990 and 1999, Missouri 1989 and 1999, and Ohio 1979 and 1993. Although data pertain only to timberland, more than 95 percent of the forest land in the Central Hardwood region is classified as timberland (Powell and others 1993). Timberland is forest land capable of producing more than 20 cubic feet per acre per year of industrial wood crops under natural conditions and not withdrawn from timber utilization.

There are other classifications of land with trees, including reserved forest land, narrow planted and natural wooded strips, and pasture land with trees that do not meet the timberland definition. These other forest land categories make important contributions to the many forest related resources and their use, but until recently, FIA did not install field plots on these other lands. As a result, data are very limited for forest land classifications other than timberland.

For stand-size class determinations, deciduous sawtimber-sized stands are stands with half or more of the total live tree stocking in trees that are at least 11.0 inches in diameter at breast height (d.b.h.). Stand age is based on the average age of the dominant and co-dominant trees in the stand. Volume is the net volume of trees at least 5.0 inches d.b.h. and over, from 1-foot above the ground (stump) to a minimum 4-inch top diameter outside bark or to the point where the central stem breaks into limbs. Live volume is used in this study, which represents the volume in live trees that include non-commercial trees, rough trees, and rotten trees on timberlands.

FIA uses a systematic sampling design with land-use of plots determined via remotely sensed data. A subset of these plots is selected for field measurement where plot level and individual tree measurements are made. All data for this study originated as field measures through the FIA program.

RESULTS

Area

Within the 5-state (Ohio, Indiana, Illinois, Missouri, and Iowa) study area, representative of the Central Hardwood region, there are currently 165.4 million acres of land, of which 32.9 million acres (20 percent) are classified as timberland (table 1). This compares to 30.6 million acres of timberland in the previous inventories and represents an 8 percent increase over the past 13+ years. While the percentage of land in trees across the Central Hardwood region is relatively small compared to other land uses such as agriculture, the impact on the landscape and the contributions from this resource are significant.

The majority of the timberland is dominated by deciduous species. Except for a few sites, conifers play a minor role in this region. Of the total area of timberland, 60 percent is currently classified as oak-hickory (*Quercus-Carya*), 19 percent as maple-basswood (*Acer-Tilia*), 13 percent as bottomland hardwoods, and 7 percent as conifers (table 1). Other forest types, such as aspen-birch (*Populus-Betula*), contribute less than 1 percent to the total area of timberland in the Central Hardwood study region. In the previous inventories, oak-hickory represented 65 percent of the total area of timberland with maple-basswood being second at 16 percent. Bottomland hardwoods represented 11 percent and conifers 7 percent in the previous inventories. Depending on the site, oak-hickory and maple-basswood forest types are considered to be self-replacing with regeneration species composition similar to their overstory. These plant communities are replacing early successional forest types such as cottonwood (*Populus deltoides* Bartr.).

The only forest type to experience a decrease in its total area between inventories is the oak-hickory forest type. All others increased with the greatest increase occurring in the

maple-basswood forest type (an increase of more than 26 percent). Bottomland hardwoods increased by 20 percent in this same time period. The increase in the maple-basswood forest type is probably due partially to newly forested acres but also to the successional change from other forest types.

More than 90 percent of all oak-hickory and maple-basswood sites in the Central Hardwood region are classified as being mesic (table 2). Timberland in the very wet (hydric) or very dry (xeric) physiographic classes is relatively rare. Less than 4 percent of the total timberland area in this region is classified as bottomlands, primarily due to the bottomland hardwood classified forest types usually being classified as either mesic or hydric (13 percent of all timberland from a forest type classification but only 4 percent of all physiographic sites).

Bottomland hardwoods are a crucial component for timber industries, recreational uses, and wildlife in the Central Hardwood region. Over time, this forestland classification has received the most pressure for conversion to agriculture and pastureland due to the productive soils associated with bottomland hardwoods. In

Table 1.—*Central Hardwood area of timberland by forest type and stand-size class**

CURRENT INVENTORIES		Stand-size class			
Forest type	Total	Large diameter	Medium diameter	Small diameter	Nonstocked
			(Thousand acres)		
Conifers	2,153.4	876.6	773.3	503.6	0.0
Oak-hickory	19,851.7	11,849.3	6,243.5	1,758.9	0.0
Bottomland hardwoods	4,135.0	2,309.2	1,046.4	779.4	0.0
Maple-basswood	6,349.2	3,266.8	1,770.1	1,312.3	0.0
Aspen-birch	124.8	14.8	73.5	36.6	0.0
Nonstocked	251.9	0.0	0.0	0.0	251.9
Total	32,866.0	18,316.7	9,906.7	4,390.7	251.9

PREVIOUS INVENTORIES		Stand-size class			
Forest type	Total	Large diameter	Medium diameter	Small diameter	Nonstocked
			(Thousand acres)		
Conifers	2,009.6	804.7	609.6	572.9	22.4
Oak-hickory	19,901.3	10,757.3	4,940.8	4,047.9	155.3
Bottomland hardwoods	3,434.1	2,068.0	682.3	659.3	24.5
Maple-basswood	5,022.8	2,496.8	998.6	1,513.5	13.9
Aspen-birch	84.4	6.9	18.1	59.4	0.0
Nonstocked	104.9	0.0	0.0	0.0	104.9
Total	30,557.1	16,133.7	7,249.4	6,853.0	321.0

* Study area of Ohio, Indiana, Illinois, Missouri, and Iowa.

Table 2.—*Central Hardwood area of timberland by forest type and physiographic class**

CURRENT INVENTORIES		Physiographic class			
Forest type	Total	Xeric	Mesic	Bottomland	Hydric
			(Thousand acres)		
Conifers	2,153.4	43.3	2,110.2	0.0	0.0
Oak-hickory	19,851.7	155.7	19,479.3	156.1	59.8
Bottomland hardwoods	4,135.0	0.0	2,949.5	858.5	326.7
Maple-basswood	6,349.2	0.0	6,264.5	67.4	17.2
Aspen-birch	124.8	0.0	124.8	0.0	0.0
Nonstocked	251.9	0.0	216.1	27.1	8.7
Total	32,866.0	198.9	31,144.4	1,109.1	412.4

* Study area of Ohio, Indiana, Illinois, Missouri, and Iowa.

general, the majority of the bottomland hardwood sites that had the potential for conversion to these land uses have been converted. However, the current pressures on forest land for conversion to other land uses derive from the demand for additional urban/suburban space, second homes, and recreational facilities. These pressures exist for both bottomland and upland hardwoods. Fortunately, we have recently seen an increase in the percentage of forest land found on bottomland sites.

Land Use Changes

When forest land is developed for other land uses its composition, structure, and function change. Natural regeneration is often curtailed; stocking is usually lowered; snags, dead trees, and hollow/rotten trees that provide excellent wildlife habitat are often removed (usually justified as a safety perspective); and other changes occur with development.

There has been a net increase in the total area of timberland since the late 1970s to early 1980s, due primarily to increases in the width of existing narrow wooded strips and the conversion of cropland and pasture to timberland. These increases are greater than losses by development of timberland for agriculture, urban/suburban expansion, and other uses. For example, in Indiana timberland increased from 3.9 million acres in 1967, to 4.2 million acres in 1986, to 4.3 million acres in 1998 (fig. 2). Between 1986 and 1998 in Indiana, 382 thousand acres of timberland were converted to nonforest land uses and 421 thousand acres of nonforest land converted to timberland. This resulted in the net increase in the total area of timberland in Indiana. The new timberland areas, however, occupy different kinds of sites than those timberland areas lost to other land uses. Historically, in the Central Hardwood region, new timberland areas have established on upland mesic sites while timberland losses have occurred on bottomlands but as noted earlier this trend may be changing.

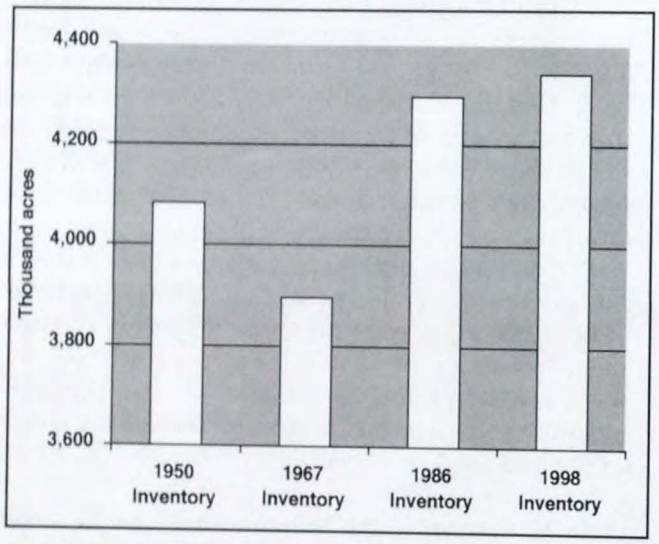

Figure 2.—*Area of timberland in Indiana by inventory time period.*

In addition to the changes in land use, changes occur in lands that remain timberland. Changes in species composition occur if the forest matures, through natural succession processes or if natural disturbances such as wind storms and floods transpire. Also, human-induced activities such as harvesting, timber stand improvement (TSI), and tree planting can change species composition. These events often result in a change in forest type classification.

Ownership

In the Central Hardwood region, 88 percent of the timberland area is privately owned (table 3). Public ownership is comprised of 8 percent in Federal, and 4 percent in State and local government ownership. Ownership by forest type generally follows the overall regional patterns for forest type. The distribution of forest land by ownership has remained relatively static over time. Although areas in public ownership have increased, the overall area of forest land has increased as well. The future of the forest resource in this region is almost totally dependent on the management objectives and intentions of private landowners.

Stand-Size Class

Larger-sized trees dominate the Central Hardwood timberland resource with 56 percent being classified in the large diameter stand-size class, or stands with dominant trees greater than or equal to 11.0 inches d.b.h. (generally referred to as sawtimber) (table 1). Medium diameter stands (poletimber) currently account for 30 percent of all timberland in the region (dominant trees between 5.0 inches and 11.0 inches d.b.h.). Small diameter (sapling-seedling) stands (dominant trees less than 5.0 inches d.b.h.) currently account for about 13 percent of the timberland area. In previous inventories, 53 percent of all timberland was classified as large diameter stands, 24 percent as medium diameter stands, and 23 percent as small diameter stands. Thus, the important change over time has been the dramatic decrease in the area of small diameter stands, a loss of more than one-third in roughly 13 years.

Table 3.—*Central Hardwood area of timberland by forest type and ownership class**

CURRENT INVENTORIES	Ownership class				
Forest type	Total	National Forest	Other federal	State and local	Private
	(Thousand acres)				
Conifers	2,153.4	316.2	62.6	130.1	1,644.6
Oak-hickory	19,851.7	1,344.2	385.2	1,083.2	17,039.1
Bottomland hardwoods	4,135.0	29.2	73.9	136.5	3,895.5
Maple-basswood	6,349.2	246.7	77.4	124.9	5,900.0
Aspen-birch	124.8	4.8	0.0	4.1	115.9
Nonstocked	251.9	28.2	0.4	1.2	222.2
Total	32,866.0	1,969.3	599.4	1,479.9	28,817.4

* Study area of Ohio, Indiana, Illinois, Missouri, and Iowa.

Average stand-size class is increasing, an indication of a lack of significant disturbance through either natural occurrences or harvesting. For example, the area of oak-hickory in the large diameter class increased by 10 percent and the area of oak-hickory in the medium diameter class increased by 26 percent between inventories. The area of oak-hickory classified as small diameter decreased by more than 57 percent during this same time period. The only forest type to increase in area of small diameter stands between inventories was the bottomland hardwoods type, showing an 18 percent increase.

Selective harvesting methods used in hardwood stands throughout the Central Hardwood region do not cause the large-scale disturbances that move larger-sized stands to smaller-sized stands. In addition, high-grading of hardwood stands tends to leave the "economically less desirable" species (such as maple, beech, and birch) and lower quality cull trees which could hasten the transition to a later successional stage.

Stocking Changes

Stocking is a measure of the degree to which land is occupied by trees. Stocking is an important concept because it influences successional processes such as the type of regeneration, individual tree growth habit, and forest structure. The goal of most forest managers is to maintain the forests in a fully stocked condition since this condition provides for full utilization of the site. However, management can be directed to obtain different stocking levels. In the Central Hardwood region, 65 percent of the timberland is currently medium to fully stocked and 9 percent is overstocked (table 4). More than 70 percent of the oak-hickory forests, compared to less than 50 percent of the bottomland hardwood forests, is currently medium to fully stocked.

Individual tree growth habitat is a reflection of stocking. Open grown trees, often referred to as "wolf" trees, generally have large well-developed crowns with larger than average branches. For the same species, open-grown trees provide considerably different resource use potential and wildlife habitat than trees grown in an overstocked condition. Depending on the species, stocking can be one of the most important criteria for determining quality of the resource from a timber management and wildlife habitat perspective.

Stand Ages

Stand age is another important characteristic describing the forest resource. Across the Central Hardwood region, 27 percent of all forest land have average stand ages of more than 80 years, considered the threshold for entering economic maturity (fig. 3). As a comparison, across the Lake States region, only 16 percent of all forest land have an average stand age of more than 80 years (Schmidt and others 1996). In the eastern United States, the Central Hardwood region has an above average amount of timberland in the older age classes, primarily due to the lack of disturbance. Mature and overmature forests provide important wildlife habitat not typical of younger-aged stands. Overstory structure, down and dead woody material, snags, and other components enable mature and overmature stands to function in a vital manner for selected wildlife species. These older forests are a source for forest products including wood fiber and fuelwood, and provide unique recreational experiences as well as soil and watershed protection.

Only 12 percent of the timberland in the Central Hardwood region has an average stand age of less than 20 years. Young stands (typically sapling-seedling stands) provide crucial habitat

Table 4.—*Central Hardwood area of timberland by forest type and stocking class**

CURRENT INVENTORIES	Stocking class					
Forest type	Total	Over	Full	Medium	Poor	Nonstocked
			(Thousand acres)			
Conifers	2,153.4	165.5	544.3	771.2	567.5	104.9
Oak-hickory	19,851.7	1,672.7	6,275.8	7,869.2	3,551.2	482.8
Bottomland hardwoods	4,135.0	340.6	751.4	1,269.6	1,264.4	509.0
Maple-basswood	6,349.2	604.0	1,459.7	2,258.1	1,635.5	391.8
Aspen-birch	124.8	20.9	26.6	25.6	51.7	0.0
Nonstocked	251.9	0.0	0.0	0.0	106.8	145.1
Total	32,866.0	2,803.7	9,057.7	12,193.8	7,177.2	1,633.6
PREVIOUS INVENTORIES						
Total	30,557.1	2,040.2	8,379.2	13,431.5	6,422.7	283.5

*Study area of Ohio, Indiana, Illinois, Missouri, and Iowa.

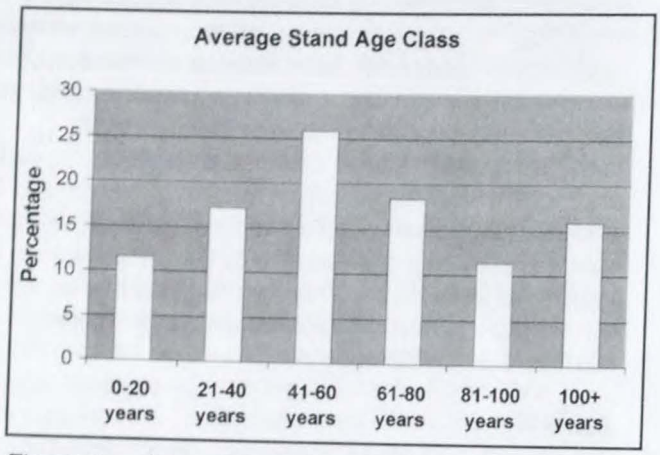

Figure 3.—*Average stand age of timberland in the Central Hardwood region.*

for many wildlife species such as white-tailed deer (*Odocoileus virginianus*) and ruffed grouse (*Bonasa umbellus*). Many wildlife species rely heavily on the seedlings, shrubs, and understory "brush" associated with these younger-aged stands. These younger stands are also a primary source of future forests; their current composition will have an important influence on the future of our forests.

Species Composition

Species composition is an important descriptor of the economic importance of timberland and its contribution to arboreal diversity and successional stage. Two states, Indiana and Missouri, are selected to quantify species composition. In the most recent inventories, a total of 92 different tree species are found on FIA plots in Indiana and 90 in Missouri. While the total number of species is similar, there are differences in the species composition. There are nine species found in Missouri but not in Indiana while, Indiana has 11 species that are not found in Missouri.

Generally, climatic changes across the Central Hardwood region result in different woody plant species compositions. As climatic factors such as precipitation, temperature, and average windspeed change, species change. The eastern portions of the Central Hardwood region typically have more species, whereas in the west where precipitation becomes more limited some species such as beech (*Fagus grandifolia* Ehrh.) and yellow-poplar (*Liriodendron tulipifera* L.) disappear from the landscape. As the species lists become smaller, the importance of the remaining species increases. On the western fringe of the Central Hardwood region in eastern Kansas and Nebraska, the oak resource is limited from the perspective of number of species and is represented by only bur oak (*Quercus macrocarpa* Michx.). The importance of bur oak in this portion of the region is of greater significance than in more easterly portions of the Central Hardwood region where more than 18 different oak species can be found.

In total, there currently are more than 17.5 billion trees in the five-state study area (table 5). Oak species are the most common, representing almost 21 percent of all live trees. The other most common species groups are maple with 11 percent, hickories with 10 percent, conifers with 5 percent, and ash (*Fraxinus* spp.) with just less than 5 percent. Black walnut (*Juglans nigra* L.) is one of the most recognized and valued species across the Central Hardwood region but only represents 1 percent of all live trees.

Less than 1 percent of the total live trees in the Central Hardwood region are at least 19.0 inches d.b.h. but these represent more than 125

Table 5.— *Number of live trees by species group and diameter class in the Central Hardwood region**

CURRENT INVENTORIES		Diameter class (in inches)			
Species group	Total	1.0-4.9	5.0-10.9	11.0-18.9	19.0+
		(Million trees)			
Conifers	887.4	610.0	238.4	38.3	0.7
Oak	3,662.9	2,227.8	972.3	399.1	63.7
Hickory	1,678.7	1,269.7	323.0	80.7	5.2
Hard maple	1,082.8	905.7	132.8	38.4	5.8
Soft maple	897.2	695.4	147.1	45.4	9.3
Ash	804.0	606.2	147.9	44.2	5.7
Black walnut	205.2	105.8	69.3	28.2	1.8
Other hardwoods	8,287.5	7,148.9	903.6	201.6	33.4
Total	17,505.7	13,569.6	2,934.5	876.0	125.5

*Study area of Ohio, Indiana, Illinois, Missouri, and Iowa

million trees. The oak species group represents 50 percent of all live trees in the region with diameters more than 19.0 inches d.b.h. Oak dominates this region, however when regeneration is considered, the continued future dominance by this species is of concern. Oaks are long-lived and, once established, not as susceptible to environmental influences such as windthrow and wildfires when compared to other species native to the study area. However, the oak resource is also threatened by oak wilt (*Ceratocystis fagacearum* (Bretz) Hunt), oak decline, and gypsy moth (*Lymantria dispar* L.) (Juzwik and Schmidt 2000). If oaks succumb to these factors, short-term wildlife habitat could be improved due to an increase in standing dead trees but long-term wildlife habitat quality could be negatively impacted if the trees are not replaced by similar species, as oaks are a critical food source for many wildlife species.

More than three-fourths of all trees are less than 5.0 inches d.b.h. The dominance of smaller sapling/seedling-sized trees could lead to the conclusion that the timberland is dominated by sapling/seedling-sized stands. However, as noted, timberland in the study area is predominately sawtimber size. The reason is that while there might be seedlings in a stand, if larger trees are present they dominate the stand and the size classification. In most timberland, the number of smaller-sized woody plants exceeds the number of larger-sized trees. The growing space needed by a seedling is dramatically different from that needed by a large dominant tree. The crucial point is that the combination of overstory and understory woody species, their interspersion, and their species composition dictate the structure and function of the forest.

The relative proportion of smaller-sized trees varies by species group. "Other hardwoods" has the greatest proportion of small trees. Currently, 86 percent of the trees in this group are less than 5.0 inches d.b.h. This group contains all of the species not associated with the other named species groups. The hard and soft maples, hickories, and ash species groups have more than 75 percent of their trees in the less than 5.0 inch size class.

The oak species group is different than any other in terms of number of smaller-sized trees. Only 60 percent of all oak trees are considered regeneration or sapling-seedlings (less than 5.0 inches d.b.h.). Oaks are of special interest due to their economic and environmental benefits. In addition, their importance as a source of food and habitat for a wide variety of wildlife species cannot be understated. For example, acorns are one of the primary plant foods for white-tailed deer (*Odocoileus virginianus*), blue jays (*Cyanocitta cristata*), and Eastern gray squirrel (*Sciurus carolinensis*). The future of oak species in the Central Hardwood region could have a direct tie to the future populations of many timber production facilities and wildlife species.

Because of the maturing of the region's forests, and the limited ability of white and red oak to regenerate under shade, the regeneration of these species is of concern. In most states, the number of oak seedlings is remaining static at best. For the near future, there should be sufficient oak regeneration to replace the existing older, mature oak following harvest or mortality of canopy trees. Nevertheless, even though oak seedlings can survive in shade for several years, they must have adequate sunlight to successfully develop. As the timberlands in the Central

Hardwood region continue to mature, future regeneration of oaks could become even more limited.

Volume

Volume has been increasing over time throughout the region. There are currently more than 44.1 billion ft^3 on the 32.9 million acres of timberland in the study region equating to almost 1,340 ft^3 per acre of timberland across the entire region (table 6). In previous inventories, total volume was 34.1 billion ft^3, an average of 1,116 ft^3 per acre throughout the Central Hardwood region. In the 13 years between inventories, total volume increased by 29 percent, continuing the trend established since the 1930s of increasing volume on the timberland in the Central Hardwood region. To provide a comparison on a local level, average volume per acre in Indiana increased from about 680 ft^3 per acre in 1950 to 1,589 ft^3 per acre in 1998.

With high volumes, concerns regarding fuel hazard, growth being suppressed due to competition, and other forest health issues increase. Greater volume levels can have either a positive or negative impact, depending on the perspective. For example, growing-stock volume in Indiana increased from 5.2 billion ft^3 in 1986 to 6.9 billion ft^3 in 1998, reflecting the increase in both area and stocking during the 12 years between those inventories. Wildlife species such as prairie warbler (*Dendroica discolor*) and yellow-breasted chat (*Icteria verens*) that prefer brushy or thicket-covered uplands could be negatively impacted from greater stocking levels, which could decrease understory "thickets" due to lower light levels (Hunter and others 2001). However, other species could be positively impacted from these greater stocking levels. From a wood fiber production perspective, the increased levels of volume per acre assist with lowering costs of accessing the timberland (road and skid trail construction for example).

In the Central Hardwood region, currently 60 percent of the total area of timberland is in the oak-hickory forest type and 51 percent of the total volume is in the oak and hickory species groups. In the previous inventories, 65 percent of the total area of timberland was in the oak-hickory forest type and 56 percent of the total volume was in the oak and hickory species

Table 6.—*All live volume by species group and diameter class in the Central Hardwood region**

CURRENT INVENTORIES		Diameter class (in inches)		
Species group	**Total**	**5.0-10.9**	**11.0-18.9**	**19.0+**
		(Million cubic feet)		
Conifers	2,026.7	1,112.0	876.2	38.5
Oak	18,468.0	4,440.4	9,222.1	4,805.5
Hickory	4,141.6	1,570.6	2,100.1	470.9
Hard maple	2,060.3	695.9	1,016.3	348.1
Soft maple	2,430.7	693.1	1,075.5	662.1
Ash	2,108.3	683.5	1,070.2	354.5
Black walnut	1,209.4	366.4	689.0	154.1
Other hardwoods	11,651.1	3,954.3	4,590.4	3,106.4
Total	44,096.1	13,516.1	20,639.9	9,940.1

PREVIOUS INVENTORIES		Diameter class (in inches)		
Species group	**Total**	**5.0-10.9**	**11.0-18.9**	**19.0+**
		(Million cubic feet)		
Conifers	1,577.8	973.4	566.7	37.7
Oak	16,103.5	4,451.9	8,086.2	3,565.5
Hickory	3,155.3	1,449.0	1,450.7	255.5
Hard maple	1,409.7	514.9	642.9	252.0
Soft maple	1,546.0	487.2	603.1	455.6
Ash	1,519.5	597.5	702.7	219.3
Black walnut	792.6	309.2	418.7	64.7
Other hardwoods	8,023.6	2,970.9	3,221.1	1,831.8
Total	34,127.8	11,754.0	15,692.0	6,682.0

*Study area of Ohio, Indiana, Illinois, Missouri, and Iowa.

groups. The lower levels of volume compared to area could imply that a number of other species exist in the oak-hickory classified timberland.

Increases in volume occurred across all species groups (table 6). While the oak species group increased its total volume between inventories by 15 percent, this was the lowest percentage increase of all species groups. Hard and soft maples, black walnut, and other hardwoods species groups all increased their total volume by more than 40 percent between inventories. In the smaller diameter size classes, the volume of oak stayed static while volume for soft maple increased 42 percent and for hard maple 35 percent. These smaller diameter classes provide a good indication of dominant timberland species 50 to 70 years in the future if there is no major disturbance. Similar trends are found for the medium diameter classes; i.e., low growth levels by oak and high growth levels for more shade tolerant species.

Change Factors

Components of change (growth, mortality, and removals) are important indicators of sustainability. Net growth is equal to gross growth minus mortality. Net growth can be compared to removals to determine the growth-to-removals ratio. A ratio of more than 1 indicates that growth is greater than removals, and hence an increase in inventory volume. Studies across the Central Hardwood region have shown that growth-to-removals ratios have been about 2.5 to 1 (Schmidt 2000). Table 7 exhibits the growth-to-removals ratio for Indiana in 1998 and table 8 exhibits the growth-to-removals ratio for an 8-state region (5 states in this study plus Pennsylvania, Tennessee, and Kentucky) conducted for an independent study (Schmidt 2000). In both cases, growth-to-removals were about 2.5 to 1. This growth-to-removals ratio indicates that each year total volume is increasing at a substantial rate, evidenced by the noted increase in total volume over time.

As would be expected, larger-sized trees have lower growth-to-removals ratios since larger trees have greater economic value from a harvesting perspective and thus are more likely to be harvested (tables 7 and 8). Typically, medium-sized trees have the greatest growth-to-removals ratios due to both the lower likelihood of being harvested and due to the physiological aspects of when the tree species in this region grow best. Growth is also high for the small diameter classes but FIA protocols don't accommodate volume estimates (total volume, growth, mortality) until the individual tree attains at least 5.0 inches in d.b.h.

Forest Structure

The distribution of trees by species and crown class (open grown, dominant, co-dominant, intermediate, and overtopped) describe general forest structure of the forest resource. Crown class distribution is a measure of the dominance of the individual tree's crown compared to the overall stand. Throughout the Central Hardwood region, a low percentage of the trees on timberland are open grown. This is primarily due to the minimum stocking level of 10 percent necessary to qualify as timberland.

From a previous study (Schmidt 2000) oaks were found to be among the most dominant species in terms of crown class distribution in a region that encompasses the Central Hardwood (table 9). Almost 80 percent of all oak trees in the region were considered to be dominant or codominant trees with intermediate, overtopped,

Table 7.—*Growth to removals ratio for Indiana in 1998*

Species group	Total	Diameter class (in inches) 5.0-10.9	11.0-18.9	19.0+
	(Growth to removals ratio)			
Conifers	17.4	13.8	23.7	0.0
Oak	1.9	7.7	2.5	1.2
Hickory	2.7	3.3	3.2	1.2
Hard maple	5.1	9.1	5.4	2.3
Soft maple	5.4	9.8	6.0	2.7
Ash	3.2	5.2	4.3	1.1
Black walnut	2.5	4.8	2.2	1.3
Other hardwoods	2.3	6.6	2.4	1.2
Total	2.6	6.6	3.0	1.3

Table 8.—*Growing-stock growth to removals ratios in the Midwestern United States**

Species group	Total	Diameter class (in inches) 5.0-8.9	9.0-14.9	15.0+
	(Growth to removals ratio)			
Conifers	1.3	2.7	1.3	0.4
Oak	1.9	4.0	2.0	1.3
Hickory	8.6	14.2	9.5	4.2
Soft maple	8.1	31.7	9.8	2.9
Hard maple	2.8	5.0	2.7	1.8
Yellow-poplar	10.5	35.4	16.8	6.5
Other hardwoods	2.3	5.1	2.3	1.3
Total	2.5	5.1	2.6	1.5

*Study area of Tennessee, Kentucky, Pennsylvania, Ohio, Indiana, Illinois, Missouri, and Iowa.

and saplings comprising the other 20 percent. As a comparison, in that study 40 percent of the sugar/black maple trees were considered intermediate or overtopped, partially representing the understory. In general, timberland with an above average percentage of shade tolerant species has a greater percentage of trees in the understory classifications.

DISCUSSION

The trends shown are expected to continue without an increase in the disturbance rate through either human-induced or natural factors. At this point, there are a number of factors that could alter the existing trends. Chipmills have been moving into the Central Hardwood region with resulting increased levels of clearcut harvesting that tend to remove more of the overstory than selective harvesting, which has been used historically. A number of oak pests have been identified that are greatly expanding their historical range (see Moltzan (2003) in these proceedings for an in-depth discussion of the impact of oak pests). If oak wilt and oak decline continue at their recent rate, the overall oak resource might be severely impacted. Discussions about allowing some wildfires to burn out on their own have been taking place and there appears to be an increase in the use of prescribed fire. These changes, and many others to come, could result in important changes in the current trends. However, at this point it appears that trends noted in this study will continue.

As documented, the Central Hardwood's timberland resource is undergoing important changes. Some of these changes are not readily apparent and are currently being masked by the current dominance of the oak resource. However, these changes do have repercussions for all interested in this resource including the timber industries and those concerned with wildlife habitat. For example, woodcock (*Philohela minor*) numbers have declined by 37 percent since 1968 in the central portions of the United States, primarily due to habitat loss and the maturing of the region's forests (Dessecker and McAuley 2001, Smith 1999).

Table 9.—*Crown class distribution by species group in the Midwestern United States**

Species group	CROWN CLASS			
	Open grown	Dominant-Codominant	Inter-mediate	Over-topped
	(Percentage)			
Conifers	0.2	64.9	32.0	2.9
Oak	0.1	78.0	20.4	1.4
Hickory	0.2	65.9	31.4	2.4
Soft maple	0.1	67.6	29.5	2.9
Hard maple	0.1	60.1	34.0	5.8
Yellow-poplar	0.2	74.2	24.1	1.5
Cottonwood-aspen	0.3	94.9	4.2	0.5
Other hardwoods	0.2	64.7	32.0	3.0
All species	0.2	69.6	27.8	2.4

*Study area of Tennessee, Kentucky, Pennsylvania, Ohio, Indiana, Illinois, Missouri, and Iowa.

The future of the resource lies in the hands of individual private landowners. Their decisions regarding management will dictate the characteristics of the future resource and its resulting products and benefits. As land ownership changes, different objectives for timberland management may emerge. Currently, it appears that some landowners are moving from a focus on wood fiber production toward a broader ecosystem approach. A portion of this change in management philosophy is due to the smaller-sized tracts that are being formed as a result of forest fragmentation. As the average size of timberland ownership decreases, the potential for management for wood fiber production decreases. This will change the resource, resulting in advantages and disadvantages, depending on your outlook.

Overstocked stands are of concern due to their increased potential for pest outbreaks, wildfire, and other threats. In an overstocked status, growth rates can be lowered, stress can be increased, and regeneration can be low due to increased competition. Both over- and understocking can be addressed through management but end results of management actions need to be weighed.

A continuing major driving force in the Central Hardwood region is the aging of timberland. With only 12 percent of the timberland having an average stand age of less than 20 years, later successional types and their associated species are replacing early successional forest types. As plant species composition changes, utilization of timberland will also change.

One reason for increased stocking and aging of the timberland is the harvesting techniques used in this region. Almost all harvesting is done through selection of individual trees for removal, which leaves the majority of the stand intact. This harvesting method does not cause disturbance (changes in light penetration, exposure of bare mineral soil, etc.) to the degree

necessary to enable early successional species and/or species that need disturbance to regenerate.

Prior to European settlement, wildfires and other natural events provided disturbances that enabled early successional forests to maintain themselves. With current management of controlling wildfires, we are allowing succession to progress. As a result, forests are maturing, woody plant species are being replaced, and volume and stocking levels are increasing. If these conditions are the desired goal no major changes are needed, however, if additional conditions are desired, changes in management strategies should be considered.

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THE STATUS OF TIMBER RESOURCES IN THE NORTH CENTRAL UNITED STATES

Neal H. Sullivan and Stephen R. Shifley[1]

ABSTRACT.—Between 1953 and 1997 the volume of standing timber in the region (growing stock) more than doubled from 37 to 83 billion cubic feet. Forests in the North Central Region grow 2.3 billion cubic feet of new wood on growing-stock trees each year. Annual removals are about half that amount. The pattern is the same in each of the seven included states (Minnesota, Wisconsin, Michigan, Indiana, Illinois, Iowa, and Missouri). At the same time, at 73 cubic feet per person, the 46 million people who live in the region consume about 3.4 billion cubic feet of timber products annually. In the North Central Region: we have 14 percent of the nation's timberland, we annually grow 10 percent of the nation's new wood (growing stock), we annually harvest 7 percent of the nation's wood (growing stock), and the people who live in the region annually consume 17 percent of the nation's wood. A failure to balance growth and harvest with consumption of forest resources in one region shifts the impacts of harvesting and production to other regions of the U.S. or other countries. It also exports the environmental and social consequences associated with the timber harvest (both the positive and the negative consequences). What are the implications for local, regional, national, and global sustainability of forests? There are many ways to alter the current balance among growth, harvest, and consumption of timber: consumers, resource managers/owners, manufacturers, and elected officials can affect the balance between forest production and forest product consumption. It is not realistic to think that in the North Central Region we could be self-sufficient in the many different kinds of forest products that we consume, but it is reasonable to consider what proportion of the total volume of wood that we consume could or should be produced within the region. In our discussions of sustainability we could begin to consider the implications of various levels of regional forest growth and harvest relative to timberland area, consumption of wood products, and forest sustainability (Shifley and Sullivan 2002).

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ROLES FOR AGROFORESTRY IN HARDWOOD REGENERATION AND NATURAL-STAND MANAGEMENT

H. E. 'Gene' Garrett[1]

ABSTRACT.—A convincing case can be made that current land-use patterns in the Central Hardwood region reflect a significant underutilization of our land-based resources. A land-use strategy is required that would allow landowners who are interested in converting marginal crop lands to forests, or unproductive woodlots to productive woodlots, to make the change without financial loss. Agroforestry has the potential to precisely do this. Agroforestry, an integrated land-use management approach for both production and conservation benefits has gained remarkable acceptance since the early 1990s. It provides the landowner the opportunity to develop a portfolio of short- and long-term investments and serves as a viable alternative to conventional forestry and agricultural practices.

Agroforestry, defined as "intensive land-use management that optimizes the benefits (physical, biological, ecological, economic, social) from biophysical interactions created when trees and/or shrubs are deliberately combined with crops and/or livestock" (Gold and others 2000), has the potential to dramatically influence the acres of new hardwood plantings and unmanaged natural forest stands placed under management. This concept that had few supporters during the 1970s and 1980s suddenly became recognized as a highly specialized and successful technology in the 1990s. In contrast to the segregation-of-use philosophy that has dominated agriculture and forestry for decades, agroforestry advocates optimal integration of trees with crops for both production and conservation benefits and differs from conventional forestry in that it provides a cash flow that is derived from companion crops grown with the trees.

OPPORTUNITIES FOR HARDWOOD AGROFORESTRY

The demand for wood products within the U.S. is projected to increase significantly during the next few decades, perhaps as much as 38 percent by the year 2050 (Powell and others 1992). Coupled with a recent history of decreased harvesting on federal lands, a unique opportunity is unfolding for agroforestry in the Central Hardwood region. The increased demands will hopefully create new markets and improved prices for hardwoods (Wiedenbeck and Aramen 1993). Since the majority of timberland in the Central Hardwood region is under nonindustrial private ownership, if landowners could be convinced that putting their stands under management is financially rewarding, not only would we see vast acreages of unmanaged forests managed, but marketing of wood products could become a significant factor in the revitalization of the family farm and rural America in general.

In addition to forested land that is in need of being placed under management, the Central Hardwood region contains millions of acres of farmland, that is either lying idle or is being conventionally farmed, that has a high erodibility index (EI). Many of these lands would better serve the owner and society alike if they were under agroforestry management. Within the U.S., more than 90 percent of farm output is produced by our largest 600,000 plus farms leaving approximately 1.5 million farms fighting for less than 10 percent of the market. Within these 1.5 million farms are millions of acres of "under used" land ideally suited for tree planting. Many of these farms are located in the Central Hardwood region and are prime candidates for hardwood tree plantings (USDA 1994, Garrett 1995).

In the Midwest alone, where considerable agroforestry research has been conducted, the five states of Missouri, Illinois, Indiana, Iowa, and Ohio, have more than 19 million acres of cropland with

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an EI greater than 10, indicating very high erodibility (Noweg and Kurtz 1987, Garrett and others 1994). Approximately 9 million of these acres are recommended for forestry plantings and would be ideally suited for agroforestry. In the Plains States of Nebraska and Kansas, an additional 16 million acres of cropland are highly erodible (EI > 8) and would lend themselves well to agroforestry practices using either pure hardwoods or hardwood/conifer mixes. Within these same 7 states, there are 5 million acres of pasture land with high potential for conversion to agroforestry and more than 10 million with medium potential for conversion (Garrett and others 1994). Conversion would not only provide increased economic returns to the landowner but also many environmental benefits while helping our nation meet its future demand for wood products.

Agroforestry management incorporates intensive practices such as tree planting, annual cultivation, fertilization, irrigation, weed control, liming, grazing animals or combinations of these and other practices. Consequently, agroforestry manipulates the agroecosystem to achieve optimal benefits. These benefits may be combinations of economic, environmental, biological, or social (Udawatta and others, in press). Optimization implies that the various factors can be combined, through compromise, to best serve the interests of the user and society. Since each user may have different objectives – soil conservation, minimization of inputs, integrated pest management, profits, aesthetics, etc. – optimization of system components will vary. However, in most agroforestry designs, protective and productive benefits are realized. They are the products of biophysical interactions resulting from the proper mix of woody perennials, herbaceous species and/or livestock. These interactions directly affect soil, water, and air quality, biological diversity, wildlife habitat, aesthetics, economics and ultimately, even rural community development.

The intentional combining of trees and/or shrubs with crops and/or livestock is an essential element of agroforestry. This aspect can create a multi-storied characteristic similar to that found in many natural ecosystems while providing a variety of marketable commercial products. Crops may include conventional agronomic commodities such as corn (*Zea mays* L.), wheat (*Triticum aestivum* L.) and soybeans (*Glycine max* (L.) Merr.); specialty commodities such as ginseng (*Panax quinquefolium* U.), mushrooms, honey and floral greens; warm- and cool-season pasture; woody biomass for energy, oriented strand board (OSB) and paper; trees of high-value timber/veneer, nut/fruit producing perennials, ornamentals and others.

AGROFORESTRY PRACTICES

Trees and crops can be arranged in numerous configurations providing five categories of agroforestry practices as recognized by the Association for Temperate Agroforestry (Merwin 1997). These practices are: Silvopasture, Forest Farming, Forested Riparian Buffers, Windbreaks, and Alley Cropping.

Agroforestry Practices For Native Stands

In the Central Hardwood region of the United States, nonindustrial private forest landowners own a high percentage of the commercial timberland. Much of this timberland is found on farms of 100 acres or less and is not under management. The reasons for lack of management are sometimes complex and highly variable. But with many, the lack of a short-term economic incentive serves as a major deterrent. In addition, forest land holdings are typically so small that, in the past, clearcutting has been perceived as one of the few viable options available to the owner to generate immediate income. With the downturn in farm economies during recent years, farm owners have become more receptive to trying agroforestry practices that allow them to conduct timber stand improvement, as a long-term investment, while creating microenvironments suitable to support forages for silvopasture or specialty crops in forest farming operations that provide an immediate cash flow.

Silvopasture

Silvopastoral practices are a form of agroforestry in which tree, forage and animal components simultaneously share a parcel of land. The forage component is typically an improved pasture of introduced grasses and/or legumes that is managed using conventional agronomic principles. Silvopasture may be simplistically described as "trees in pastures" and is the most common form of agroforestry practiced in developed countries (Sharrow 1999). It differs from forest grazing in which grazed forests or woodlands are *extensively* managed as pseudo-natural ecosystems. While they provide wood products and some forage for livestock, as is the case with silvopastures, the opportunity for maximizing economic gain is not realized and often decreases as a result of damage to the natural ecosystem that is trying to be maintained. Silvopasture is an artificial agroecosystem where each component (i.e., tree, forage,

and animal) is emphasized and is maintained using *intensive* approaches such as annual inputs of nutrients, herbicides, and mechanical treatments (Clason and Sharrow 2000). One of the primary goals of silvopasture is to maximize economic gain on a per acre basis even though some sacrifice in wood, forage, or animal yields, from that of a pure practice, may occur.

While silvopasturing has been researched most extensively in the southern pine region, the practice has great potential within the Central Hardwood region. Even though density of shade is typically greater for hardwoods than pines, through the selection of forage species that will tolerate some shade and management of light regimes within a stand through well-timed thinnings and prunings, high quality pastures can be created and maintained under hardwood stands. Research within the Center for Agroforestry at the University of Missouri has clearly demonstrated that some forages possess the potential to grow as well in 50 percent shade as in full sun (table 1, Lin and others 1999, 2001).

Of 30 forage species and cultivars studied, 15 species/cultivars were found to be suitable based upon shade tolerance for use as a companion crop during the establishment or developmental stages of a silvopastoral practice. Those found to have slight to moderate shade tolerance such as Kentucky bluegrass (*Poa pratensis* L.), 'Benchmark' orchardgrass (*Dactylis glomerata* L.), 'Manhattan II' ryegrass

Table 1.— *Mean total above ground dry weight, acid detergent fiber (ADF), neutral detergent fiber (NDF) and crude protein (CP) percent of forages under three levels of shade during the 1994 and 1995 summer-fall growing season (Lin and others 1999, 2001)*

MEASURED PARAMETER AND SPECIES	Shade levels		
	Full sun	50% sun	20% sun
DRY WEIGHT	**- g -**	**- g -**	**- g -**
Kentucky bluegrass	12.45 a*	12.30 a	8.06 b
Orchardgrass 'Benchmark'	13.83 a	11.73 a	6.36 b
Ryegrass 'Manhattan II'	12.69 a	11.10 ab	8.59 b
Smooth bromegrass	9.61 b	11.95 a	9.54 b
Tall Fescue 'KY31'	13.28 a	16.24 a	7.96 b
Timothy	10.23 a	8.97 a	5.49 b
CRUDE PROTEIN	**- % -**	**- % -**	**- % -**
Kentucky bluegrass	20.3 b	20.7 b	22.7 a
Orchardgrass 'Benchmark'	12.6 c	15.7 b	19.6 a
Ryegrass 'Manhattan II'	15.3 b	16.0 b	18.8 a
Smooth bromegrass	16.7 c	18.1 b	20.2 a
Tall Fescue 'KY31'	14.0 b	15.0 b	18.1 a
Timothy	15.4 c	17.6 b	20.4 a
ACID DETERGENT FIBER	**- % -**	**- % -**	**- % -**
Kentucky bluegrass	33.0 b	35.3 a	33.9 ab
Orchard 'Benchmark'	35.2 a	35.8 a	31.5 b
Ryegrass 'Manhattan II'	31.5 a	32.5 a	32.3 a
Smooth bromegrass	33.0 a	34.0 a	34.2 a
Tall fescue 'KY31'	32.9 b	35.1 a	33.0 b
Timothy	32.4 b	34.1 ab	35.6 a
NEUTRAL DETERGENT FIBER	**- % -**	**- % -**	**- % -**
Kentucky bluegrass	60.9 a	64.1 a	62.7 a
Orchard 'Benchmark'	65.0 a	63.9 a	61.3 a
Ryegrass 'Manhattan II'	59.6 a	59.2 a	56.4 a
Smooth bromegrass	54.8 b	58.0 a	55.2 ab
Tall fescue 'KY31'	62.1 a	61.8 a	59.8 a
Timothy	58.6 a	60.7 a	58.4 a

* Means followed by the same letter within a row do not differ significantly from each other at the 5 percent level of probability using Tukey's studentized range test.

(*Lolium perenne* L.), and timothy (*Phileum pretense* L.) are recommended for use during tree establishment, under conditions of low density tree arrangements, or with tree species having open canopy structures at maturity. Because of the competitive nature of fescue (*Festuca arundinacea* Schreb.), it is recommended only for well-established stands. Others that demonstrated high shade tolerance, such as smooth bromegrass (*Bromus inermis* Leyss.) are recommended for incorporation into silvopastures containing tree species with more dense canopies or into mature silvopastures (Lin and others 1999). Even with the more shade tolerant forages, yields will be a product of shade levels. Therefore, canopy management through appropriately timed thinnings and prunings is always important.

More recent work within the University of Missouri's Center for Agroforestry has demonstrated that while percent acid detergent fiber (ADF) and neutral detergent fiber (NDF) are either not affected or only slightly increased by shade, overall forage quality for some species is increased (Lin and others 2001, Huck and others 2002). This is in part attributed to shade creating a concentration of nitrogenous compounds in the foliage. In Lin and Huck's work, crude protein was found to increase in shade tolerant forages when grown under light shade. Furthermore, Huck and others (2002) found increased digestibility of some forages when they were grown in partial shade. Grazing trials in Missouri using mixtures of these forages and others have proven successful (Lehmkuhler and others 1999).

Forest Farming

Forested areas also provide opportunities for the production of specialty crops that are sold for ornamental, culinary, or medicinal uses. Forest landowners not interested in livestock grazing may choose to develop enterprises such as the production of food items (mushrooms, maple products), medicinal plants (ginseng, goldenseal (*Hydrastis canadensis* L.)) or even shade-tolerant ornamental plants. Forest farming requires thinning of natural stands to create suitable microenvironments for growing specialty crops such as shade-tolerant botanicals. Specialty crops serve to provide a cash-flow while the released trees (the long-term investment) are growing to a commercial size. The sale of medicinal botanicals (just one of many options available) has reached the 4 billion dollar mark in the United States and represents one of the fastest growing segments of mass marketing (supermarket, drug, and natural food stores—Brun 1999).

For landowners who feel they must have short-term economic justification for placing their stands under management, forest farming provides this justification. By carefully matching a commercially valuable companion crop to the site and microenvironment created, the landowner can realize significantly greater economic gain than is possible from conventional forestry practices alone.

In the hardwood region of the central and eastern United States, forest farming first took on prominence in the northeastern and Appalachian States but more recently has attracted interest in the Corn Belt region. Its popularity is rapidly growing as woodland owners become more aware of the economic aspects of special forest products. Naturally forested areas throughout the hardwood region provide excellent opportunities for the production of specialty crops. Even though currently most forest specialty crop production is passive (i.e., harvested as a wild crop) and does not qualify as agroforestry using the strict definition, the potential for increasing the economic gain through intensive management is rapidly becoming recognized as markets develop for plants common to understories.

Agroforestry and Tree Plantings

Many landowners with underutilized acreage are expressing interest in planting single or multiple rows of trees in agroforestry configurations to provide production and environmental enhancement benefits. Agroforestry provides opportunities through the application of silvopasture, alley cropping, windbreaks, and forested riparian buffer practices.

Properly designed agroforestry practices, regardless of the one chosen, can yield many benefits but their success requires proper selection and active manipulation of the vegetation. While the potential for agroforestry varies by regions, reflecting the diverse landscapes, values, and regional/local economies, the ultimate success of an agroforestry program depends upon the nature of the biological interactions created and the value of the benefits (economic, environmental, etc.) produced.

In agroforestry practices started from planting, selection of the proper tree species is of paramount importance. In addition to being adapted to the site and producing high-value products

(i.e., wood, fruit, specialty products etc.), the trees must create suitable microenvironments to accommodate the needs of companion crops while satisfying special needs such as controlling erosion, improving wildlife habitat, or buffering our waterways from excess nutrients and pesticides. While the emphasis on desirable characteristics of hardwood trees used in agroforestry will vary to some degree with management objectives, there is a consensus that certain characteristics are generally desirable. Desirable characteristics might include the production of multiple high-value products (i.e., wood, fruit, specialty products, etc.); generation of light shade; deep-rootedness with minimal surface roots (except where erosion control is a priority); rapidly decomposing foliage; foliage that minimizes acid-generating potential; a lack of allelochemicals; a short growing season; and, good wildlife food and/or habitat production potential. Tree spacing and the most effective design will vary depending on the management practice and purpose of the planting, light requirements of the species used (including companion crops), and even personal preferences of the landowner.

Silvopasture

While silvopasture can serve to get unmanaged native stands under management, it is also a viable option for new plantings. In recently established and young open-grown stands, light is abundant and most grasses and legumes will produce high quality forage for pasture. However, to be successful, a landowner must provide some form of protection for the trees. Placement of a wire cage around each seedling is effective but expensive. An equally effective and more preferred practice uses a single strand of electric wire down both sides of a row of trees. Lehmkuhler and others (2002) found no damage to four hardwood species planted directly into pastures when protected by electric fencing. A landowner might also choose to allow the planted trees to reach a suitable size before pasturing the area, thus saving the cost of protecting the trees.

To avoid the negative effects of grazing cattle, including compaction, rotational rather than continuous grazing is recommended in silvopastures. Lehmkuhler and others (1999), found that 20-year-old black walnut (*Juglans nigra* L.) plantations placed under silvopastoral management, produced adequate quantities of forage to maintain cattle during the spring and early summer months utilizing cool-season grasses and legumes (table 2). Rotationally grazing the plantations, compared with continuous grazing, provided increased forage production and quality, higher calf gains and less soil disturbance without negatively affecting tree growth.

Alley Cropping

Alley cropping is broadly defined as the planting of rows of trees at wide spacings that create alleyways within which agricultural or horticultural crops are grown (Garrett and McGraw 2000). In the Midwest where this practice is most popular, high-value hardwoods are often used to create alleys that support conventional row, forage, or horticultural crops. Within- and between-row spacings vary with the tree and intercrop species planted, management objectives (i.e., emphasis on wood versus fruit production) and even the availability of farming equipment (width of headers, etc.).

Table 2.—*Weekly and overall average total and live forage dry matter (DM) availability from silvopastures in southwest Missouri, USA (Lehmkuhler and others 1999)*

Week	Total DM (lb acre^{-1})			Live DM (lb acre^{-1})		
	Continuous	Rotation	SEM	Continuous	Rotation	SEM
1	1,668[a]	2,581[b]	160	1,632[a]	2,513[b]	149
2	1,042	1,398	160	1,026	1,386	149
3	933	1,491	160	796	1,341	149
4	424	760	160	279	533	149
5	513	808	160	246	505	149
6	598	614	160	218	376	149
7	217	363	160	93	190	149
8	125	469	160	24	355	149
Overall mean	698[a]	1,060[b]	56	539[a]	900[b]	53

[a,b] Treatment means within a row with unlike superscripts differ significantly ($P < 0.05$).

While few reports are available on the effects of the open conditions in alley cropping on tree growth and wood quality, those available are favorable. Cutter and Garrett (1993) found that height, diameter, and specific gravity were equal to or better for alley-cropped walnut trees than for trees grown under woodlot conditions. Furthermore, based upon early growth of alley-cropped walnuts, 50- to 60-year veneer and saw log rotations were projected compared with more standard 80- to 100-year rotations for more conventionally grown trees of higher density (Garrett and others 1995). Assuming equal quality, the shorter the rotation, the greater the return on the investment. Although published information is limited, that which is currently available suggests that if properly managed, the quality of hardwood trees grown under alley-cropping, will not vary significantly from that of forest-grown trees but the time required to grow them to a marketable size will be greatly reduced. Greater management (i.e., pruning, etc.), however, is required to guarantee high quality logs with alley-cropped trees than with trees grown at higher densities.

Various economic analyses have been conducted on walnut alley cropping in Missouri (Garrett and others 1994; Kurtz and others 1984, 1996). Depending upon the combination of crops studied, internal rates of return (IRR) for walnut alley cropping have been found to range from 4 to 11 percent. In general, returns tend to increase with management complexity and site quality. However, obvious factors such as market value of crops grown, cash-flow relationships and even risk taking, influence profitability. With black walnut, early returns from planted intercrops coupled with nut production revenue are very important factors in creating high financial yields (Garrett and Harper 1999, Kurtz and others 1996). Because of this and the uncertainty of good nut production resulting from wild, genetically unknown walnut seedlings (Garrett and others 1995, Jones and others 1994), the planting of grafted walnut cultivars with known genetic histories is strongly encouraged. Similarly, if individuals are interested in using pecan (*Carya illinoensis* (Wanyenh.) K. Koch), Chinese chestnut (*Castanea mollissima* Bl.) or some other nut or fruit-bearing species in alley cropping or other agroforestry practices, they should give serious consideration to using reliable grafted selections.

Windbreaks

Windbreaks are of noted importance in the Plains States for protecting and enhancing production of crops (Brandle and others 1988). However, with our increased knowledge of their value, has come increased adoption and use in other areas of the Central Hardwood region. When properly designed, woody/herbaceous windbreaks can be used to protect soil, conserve moisture and improve crop and animal production. They also provide benefits to wildlife by protecting them from wind and adverse weather, and provide escape or refuge cover, food and foraging sites, reproductive habitat and travel areas (Pierce and others in press, Johnson and others 1993).

Windbreaks also provide diversity that supports natural enemies of crop pests and are finding increased use around animal confinements for odor regulation. Twenty-year-old windbreaks in which trees and shrubs occupied only 6 percent of the land unit have been shown to provide adequate crop protection in Nebraska (Brandle and others 1988). Increases in yields from windbreaks vary with the crop and year (weather). However, increases in winter wheat will typically be in the 20+ percent range and some crops like millet (*Tennisetum glaucum* L.) may yield 40 or more percent higher behind windbreaks than in the open (table 3, Brandle and others 2000).

Windbreaks can also be beneficial for livestock in the more northern areas of the Central Hardwood region. Properly positioned hardwoods or conifers combined with hardwoods can provide much needed protection for pastures, feedlots, and calving areas. Reducing wind speed lowers animal stress, improves animal health, and increases feeding efficiency of livestock (Brandle and others 2000). Canadian researchers have demonstrated that cattle on winter range require an additional 20 percent increase in feed energy, above maintenance, to

Table 3.—*Crop response to shelter (Brandle and others 2000)*

Crop	Number of field years	Weighted mean yield increase
	- no. -	- % -
Winter wheat	131	23
Barley	30	25
Oat	48	6
Millet	18	44
Corn	209	12
Soybean	17	15

offset the direct effects of exposure to a combination of cold temperatures and wind. Adequate wind protection has been found to reduce the direct effects of cold by more than half (Webster 1970). Similar findings have been reported for other animals such as dairy cows and sheep (Brandle and others 2000).

While some negative effects of windbreaks exist, most can be mitigated through proper species selection, corridor design, and management. Effects perceived as being negative by landowners include a reduction in available land for planting, the potential for increased wind turbulence, competition for soil moisture and nutrients, allelopathy, and shading. All of these potentially negative effects occur in immediate proximity to the woody windbreak and are manageable.

The economic benefits of windbreaks are quite variable and location specific due to the indirect nature of windbreak contributions. While numerous studies have been published describing the specific gains in increased crop and livestock yields, few financial evaluations of the costs associated with generating the gains are available. However, a detailed examination of the financial aspects of several field windbreak practices over a range of crops, yields, prices, discount rates, and windbreak establishment costs has been conducted for eastern Nebraska. Three practices utilizing windbreaks at spacings of 635, 420, and 218 feet were evaluated in fields planted to soybeans, corn, or winter wheat. All three designs were found to yield positive net benefits over an estimated 50-year life span. The lowest return was from the widest spacing, although a positive present net worth (PNW) at an 11 percent discount rate was still realized. Positive PNW's were yielded by the other designs at discount rates as high as 17 percent (Brandle and others 1992). Similarly, positive economic returns have been reported for windbreaks designed to protect animals (Quam and others 1994).

Forested Riparian Buffers

An agroforestry practice of enormous potential in the Central Hardwood region and in need of immediate broadscale adoption is the forested riparian buffer. This practice typically consists of a combination of herbaceous and woody species established on stream and river banks and is designed to help control water quality, flow regime, physical habitat, and energy inputs in streams, while producing wood and other products of commercial value.

Agriculture-derived contaminants such as sediment, nutrients, and pesticides, constitute the largest diffuse source of water-quality degradation in the Central Hardwood region. Surface runoff and subsurface flow from pastures, crop fields, and animal confinements can cause significant nutrient loading to downslope water sources unless appropriate management techniques are applied. Bioassimilative strategies using properly designed forested riparian buffers are often advocated. Research has demonstrated that inclusion of trees and shrubs with grasses can be an effective tool in countering water pollution problems especially when they are established along smaller headwater streams (Osborne and Wiley 1988).

Woody/herbaceous riparian buffers function as bioassimilative transformers, changing the chemical composition of compounds (Lowrance 1992, Lin and others 2002). Under oxygenated soil conditions, resident bacteria and fungi mineralize runoff-derived nitrogen and other compounds that are then available for uptake by soil bacteria and plants. In addition, greater infiltration of contaminant-transporting water occurs within the forested riparian strip than in cultivated fields and pastures (Schultz and others 2000). Planting or retaining forested riparian buffers are effective and economically feasible procedures for reducing chemical contaminants. Established forested riparian buffers 90- to 150-feet in width have been shown to reduce nitrogen in ground water by 68- to 100-percent and in surface runoff by 78- to 98-percent. Buffers 50- to 150-feet wide reduce phosphorus concentrations in surface waters by 50- to 85-percent (Osborne and Wiley 1988).

Sediment loading of streams can also be a problem but is even easier to address with forest/grass buffers than is nutrient loading. Narrow buffers—30- to 90-feet wide—that augment shallow sheet flows provide excellent protection from sediments entering streams (Osborne and Kovacic 1993). While stiff, dense grass is an important component of the buffer to drop-out sediment contained in runoff, the woody component is necessary to stabilize streambanks. Sloughing streambanks are one of the greatest sources for sediment found in our streams (Erman and others 1977).

Narrow wooded buffers have also been shown to have a significant effect on reducing solar radiation inputs and effectively moderating stream water temperatures. Feller (1981) demonstrated

that 30- to 90-feet wide buffers improved habitat for many aquatic insects via reduced stream water temperature, higher dissolved oxygen and reduced denitrification rates. Partial shade is required when attempting to maximize both the overall ecological health of the riparian environment and control specific submerged and emergent aquatic plants.

Little information exists related to the time required, or the specific composition of forest riparian buffers needed, to restore farm streams to their former undegraded condition. However, the requirement and time may be less than previously thought. In 1985, a woody riparian buffer was established in southern Ontario using hardwood species common to the Central Hardwood region including cottonwood (*Populus deltoides* Bartr.) and river birch (*Betula nigra* L.). A 1.2-mile section of a small degraded stream was planted on both sides with 6,500 trees. After only 4 years, the plantings averaged 8 dried tons of biomass per acre and radiation loading to the middle of the stream was reduced by almost 40 percent (Gordon and others 1992).

In recent work within the Center for Agroforestry at the University of Missouri, Udawatta and others (in press), found that after only 3 years grass buffers planted with single rows of oak (*Quercus* spp.) and integrated into an upland site, reduced total phosphorus in runoff by 17 percent based on calibration relationships. In view of the miles of unprotected stream and river banks in the Central Hardwood region—the states of Illinois, Indiana, Iowa, Missouri, and Ohio have 85,000 miles (Garrett and others 1994)—there are many opportunities for planting of hardwoods for both environmental and economic benefits. Economic gain from riparian buffers varies with design. Revenues are associated with marketable products such as forage, fiber, specialty crops, and timber produced within the buffer (Schultz and others 2000).

CONCLUDING REMARKS

Never in history has global concern for the consequences of human land use been more widely shared. Soil erosion adds to food, energy, and transportation costs and threatens future food production capacity. Nonpoint source pollution from forest and agricultural lands restricts access to safe water. Loss of vegetation from land development and site degradation affects our aesthetic environment, global climate patterns, and the quality of the air we breathe. Such problems are often the legacy of our success in maximizing production of one or more agricultural products in a financially optimal fashion without sufficient knowledge of, or regard for, impacts on future productivity and the environment. This was a rational economic choice when long-term consequences were unknown, when resources appeared to be relatively unlimited, and when technology promised means for further intensifying production.

Now that we are discovering undesirable longer-term consequences of current land-use systems, alternatives must be sought. One alternative is to model managed ecosystems after the structure and functions of naturally occurring ecosystems by re-establishing complexity in time, space, and biodiversity. This would lead to a shift away from separating land uses on discrete parcels to integrating them on a landscape level. Agroforestry, which exploits the interactions between trees and crops (including livestock) when they are grown together, bridges the gap between production agriculture and natural resource management. This provides opportunities to integrate land uses on a landscape level. Furthermore, properly designed agroforestry practices provide environmentally and economically sound alternatives to many unsustainable production systems in use today.

Agroforestry seeks to optimize production of multiple products and benefits by manipulating the interactions between components. It requires shifting our thinking in both spatial and temporal domains, and demands skills in managing, rather than reducing complexity. Traditional disciplinary approaches to problem-solving are no longer sufficient. Agroforestry challenges land managers to transcend disciplinary boundaries and explore the potential synergism between production agriculture and natural resource management. Essential to this is an understanding of hierarchical scalar relationships within ecosystems and recognition that defined ecosystem "boundaries" exist primarily for managerial convenience.

Two decades of observational data and applied research suggest that agroforestry should be vigorously explored as a possible component of improved land-use strategies in the Central Hardwood region and throughout the United States. Current interest in ecosystem management strongly suggests that we should embrace the complexity inherent to agroforestry and apply agroforestry principles, where appropriate, to better meet our current and future needs for the products and services of the land.

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THE FIRE AND OAK HYPOTHESIS: INCORPORATING THE INFLUENCE OF DEER BROWSING AND CANOPY GAPS

Rachel J. Collins and Walter P. Carson[1]

ABSTRACT.—A century of fire suppression has altered tree species composition and is a commonly cited cause for the region-wide decline in oak abundance (the fire and oak hypothesis). Other explanations include alterations in canopy gap regimes and deer browsing that operate in conjunction with fire suppression. We examined the interactions among these processes by manipulating fire, deer browsing, and canopy gaps, in a fully factorial design. Fire lowered survival of small canopy trees (10-19.9 cm DBH) but had no effect on large canopy trees (≥ 20 cm DBH). Fire increased and deer browsing decreased the proportion of top-killed saplings that sprouted. Gaps, however, had no significant effect on sprouting. Deer browsing, after fire, reduced diversity in the sprouting community, created understories dominated by striped maple. Northern red oak saplings were not fire tolerant and did not produce tall sprouts following fire. These results cast doubt on the ubiquitous application of the fire and oak hypothesis to explain the dominance of oak in some mixed hardwood forests.

Historically, re-occurring fire may have caused oak to be common across much of the eastern deciduous forest: The fire and oak hypothesis (Abrams 1992). Proponents of the fire and oak hypothesis argue that a century of fire suppression is tied not only to a decline in the abundance of oak but a concomitant increase in the abundance of maple and other mesic species (Crow 1988, Brose and others 2001). Fire could promote oak if oaks survived fire in greater proportions than co-occurring species (Lorimer 1985). Oaks may increase in relative abundance following fire either via lower stem mortality or a greater capacity for sprouting or both (Crow 1988, Abrams 1992, Van Lear and Watt 1992). Both the ability of trees to survive fire and sprout following fire varies widely among species (Wright and Bailey 1982). Re-occurring fire may have favored sprouting as a major pathway of regeneration because it skips the vulnerable early establishment phase (Bond and Midgley 2001). Overall, fire can structure forest communities and set successional trajectories through species' differential tolerances to fire.

The abundance of oak is typically correlated with fire frequency. For example, frequent, re-occurring fires over the past 10,000 years are associated with periods of high oak abundance (Delcourt and Delcourt 1987). Guyette and Dey (1996) and Abrams (1992) argued that logging, circa 1900, and subsequent fires expanded the dominance of oak into areas (e.g., Ontario and Wisconsin) that were formerly dominated by more mesic species such as beech and maple. In the last 50 years, more mesic and putatively fire intolerant species have invaded the understory of mature oak forests, where fires have been largely suppressed for a hundred years (Lorimer 1984). Although these studies provide strong support for a role of fire in oak forest dynamics, the evidence remains circumstantial and correlative.

Some prescribed fire studies have supported the fire and oak hypothesis particularly when applied to northern red oak. Canopy oaks typically have high survival following prescribed fire of low to intermediate intensity (Elliott and others 1999) and often survive in higher proportions relative to some

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co-occurring species (Brose and Van Lear 1999). Understory oaks vary in their response to prescribed fires. In some cases, oak seedlings and saplings sprout in higher proportions, have greater survival, and have greater growth rates following fire compared to co-occurring species (Christensen 1971, Huddle and Pallardy 1996, Brose and Van Lear 1998). For example, in Wisconsin, small prescribed fires increased the relative densities of oak seedlings in forest openings (Kruger and Reich 1997). In contrast, other researchers have found that the abundance of northern red oak seedlings decreased after fires (Wendel and Smith 1986, Merritt and Pope 1991, Elliott and others 1999).

Because site-specific factors likely interact with the effects of surface fire on tree species survival, it is not surprising that little consensus about the effects of fire on oak has emerged from a handful of prescribed fire experiments. For example, deer browsing and canopy gaps strongly favor some species over others and the degree to which these factors operate varies greatly across forest types and landscapes (Ehrenfeld 1980, Marquis 1981, Runkle 1990, Castleberry and others 2000). How these two factors interact with fire remains little more than speculation but may be central to an understanding of forest dynamics in the eastern USA.

Deer herbivory can dramatically alter species composition in the seedling and sapling layer due to both interspecific differences in palatability and differential survival and regrowth following browsing (Anderson and Loucks 1979, Tilghman 1989, Ford and others 1994, Kittredge and others 1995). Further, deer may preferentially browse in burned areas due to an increase in forage (Masters and others 1993). For example, Gordon and others (1995) found that oak seedling growth was lower in burned sites and attributed this to intense browsing by deer. Studying fire without explicitly considering the impact of browsing could lead to spurious conclusions.

Similar to deer browsing, canopy gaps can alter species composition in the understory due to interspecific differences in shade tolerance among seedlings and saplings (Ehrenfeld 1980). In canopy gaps, species that are intermediate in shade tolerance, such as northern red oak, may be able to regenerate (Kline and Cottam 1979, Barden 1981). In fact, Nyland and others (1982) found that oak saplings were present almost entirely in canopy gaps. Furthermore, Runkle (1990) demonstrated that canopy gaps permit some species to regenerate in the absence of stand replacement disturbance and thus can maintain tree species diversity in forests. Additionally, deer may preferentially browse on abundant sprouts found in forest openings compared to closed canopy areas (Butterworth and Tzilkowski 1990, Castleberry and others 2000). Because interactions are important and understudied, we argue that it is time to move beyond hypotheses that narrowly focus on a single factor (e.g., the Fire and Oak Hypothesis) when the influence of other factors are well demonstrated.

Overall, there is ample evidence to argue that fire, canopy gaps, and browsing will all interact to strongly influence the diversity and relative abundance of understory trees. Here, we experimentally evaluated the influence of canopy gaps, deer browsing, and fire on canopy survival and sapling sprouting of 14 common tree species using a factorial design (see Appendix table 1 for plant species code, scientific, and common names). In particular, we tested the predictions of the fire and oak hypothesis that

1) a greater proportion of northern red oak canopy trees will survive fire than co-occurring species, and
2) after prescribed fires, a greater proportion of top-killed northern red oak saplings will resprout than co-occurring species.

We go on to expand the fire hypothesis by explicitly considering the potential interactions of deer browsing and canopy gaps on a community of co-occurring species.

METHODS

Study Area and Design

We conducted this study in four 20 to 40 ha stands at the Westvaco Research Forest (WRF) in Randolph County, in north-central West Virginia. The WRF encompasses 3,500 ha and lies within the mixed mesophytic forest region (Cumberland and Allegheny Plateaus) of the eastern deciduous forest (Braun 1967). The four stands had slopes from 10 to 31 percent, were between 800 to 1,000 m in elevation, and ranged in age from 64 to 77 years. Basal area for these stands was between 22 and 28 m^2/ha with site indexes ranging from 21-24 m (70-80 ft for northern red oak at 50 years). We chose stands that had not been logged or thinned in more than 15 years, that had a closed canopy, and were devoid of signs of recent major disturbance including large canopy gaps and fire. Although sugar maple dominated both the

canopy and sapling layers, the stands were relatively diverse and typical of mixed mesophytic/ northern hardwood transition forests (table 1).

The study was installed as a split-plot design. Within each of the four stands, we established 16, 400 m^2 plots (20 by 20 m) in 1997. All plots were located at least 20 m from other plots, stand edges, and fire breaks. We avoided canopy gaps and large rock outcroppings. Half of each stand, including 8 of the 16 plots, was randomly assigned to a burned treatment or an unburned control treatment. Within each half, each of the following four treatments, fence (no deer browsing), canopy gap (250 m^2), fence x canopy gap, and neither a fence nor gap, were randomly applied to two of the eight plots.

Treatments

We created 32 canopy gaps (~18 m in diameter, 250 m^2) by girdling and injecting herbicide (Accord® by Monsanto) into trees (DBH > 10 cm) located in the center of each plot in the spring of 1998 and erected deer proof fences around 32 plots immediately thereafter. We excluded deer with 2 m tall fences constructed from 14 gauge farm and field woven wire fencing reinforced with iron rods (rebar) and metal posts erected in the summer of 1998. These fences primarily excluded white-tail deer (*Odocoileus virginianus*). The mesh size was approximately 30 x 15 cm allowing small to medium sized animals (e.g., raccoons) to pass through and other animals could fly or climb over (e.g., turkey and black bear). We found no deer hoof prints during snow cover or deer scat in any season inside fences. We refer to the fence treatment as "no-deer" for the remainder of the paper.

Four 10 to 20 ha fires were conducted between 30 April and 4 May 1999 after 6 to 10 days of no rain. The fires occurred during the sapling layer bud break, which we observed occurs 1 to 2 weeks prior to canopy layer bud break. Fuel loads ranged from 237 to 325 kg/ha of leaf litter and small woody debris (≤ 3 cm diameter, dry weight directly sampled in 192, 0.25 m^2 quadrates) and 1,450 to 2,540 kg/ha of course woody debris (estimated). Air temperature ranged from 20° to 23° C, relative humidity ranged from 20 to 28 percent, winds ranged from 0 to 13 kph. Fires were started from late morning to mid afternoon using strip head fires. Strips were set from 20 to 60 m apart using drip torches. Fires were hot and mean temperatures per fire ranged from 215° to 440° C at the ground level measured with 128 temperature sensors made from Tempilax® temperature sensitive liquid following the procedure described by Cole and others (1992). Percent of ground burned per fire ranged from 78 to 100 percent.

Plot Census Data

In 1997, the year prior to commencing treatments, we identified, permanently tagged, mapped, and

Table 1.—*Species composition in the canopy layer (≥ 10 cm DBH) and sapling layer (10-9.9 cm DBH). Density (stems/ha), SE and percent of stand are means of the four stands*

Species	Density	SE	Percent
CANOPY LAYER			
Sugar maple	223.05	55.70	42.6
American beech	66.02	40.59	12.6
Red maple	52.73	17.52	10.1
Basswood	42.58	15.41	8.1
Magnolia	27.34	19.12	5.2
Birch	20.70	7.93	4.0
Northern red oak	20.31	7.99	3.9
Black cherry	19.14	5.93	3.7
Yellow poplar	15.23	6.33	2.9
White ash	14.45	2.81	2.8
Hickory	6.25	3.55	1.2
Chestnut oak	4.30	4.30	0.8
Sourwood	3.91	3.41	0.7
Black locust	3.13	0.64	0.6
Eastern hemlock	2.73	2.24	0.5
Ironwood	1.56	0.64	0.3
Striped maple	0		0.0
Witch hazel	0		0.0
Total	525.39	27.03	
SAPLING LAYER			
Sugar maple	1,442.97	67.06	47.3
American beech	544.92	325.53	17.9
Red maple	90.63	66.53	3.0
Basswood	18.36	5.79	0.6
Magnolia	53.52	29.05	1.8
Birch	81.64	28.32	2.7
Northern red oak	14.84	8.80	0.5
Black cherry	218.36	81.18	7.2
Yellow poplar	10.55	3.15	0.3
White ash	16.02	9.79	0.5
Hickory	11.33	7.14	0.4
Chestnut oak	0		0.0
Sourwood	0		0.0
Black locust	0		0.0
Eastern hemlock	0		0.0
Ironwood	38.67	5.68	1.3
Striped maple	458.98	226.55	15.1
Witch hazel	48.83	33.01	1.6
Total	2,941.06	548.69	

recorded DBH for all trees greater than 140 cm tall for a total of 636 canopy trees (> 10 cm DBH) and 7,905 saplings (1-10 cm DBH). We remeasured all trees in all plots in June and July 1999 and 2000. Trees were designated as alive or dead. Saplings were designated as alive, top-killed with sprouts, or top-killed without sprouts. We also quantified the number of sprouts on all top-killed saplings and measured the height of the tallest sprout. We used "top-killed" to describe saplings where the original aboveground biomass was dead when we remeasured. We used "top-killed with sprouts" to describe top-killed saplings that had new shoots originating from near the root collar during the recensuses.

Canopy Tree Survival

We analyzed survival among treatments in two size classes of canopy trees (small: 10-19.9 cm; and large: ≥ 20 cm DBH) on a per plot basis for the 10 most common species (table 1). Because we girdled canopy trees in gap plots, we excluded gap treatments from all survival analyses; this included plots where gaps were crossed with other treatments (i.e., NO-DEER X GAP, FIRE X GAP, FIRE X NO-DEER X GAP). Because percent survival for many species and plots was close to 100 percent, we performed an arcsine transformation on all of the proportional survival data to normalize the distributions (Sokal and Rohlf 1995). We tested mean survival for all species combined with an ANOVA using a split-plot, two-way factorial design with FIRE as the main factor and NO-DEER, canopy tree SIZE, and NO-DEER X SIZE as within plot factors. For this and all following analyses that included FIRE as a factor, the F-statistics for the main-plot effect of FIRE was calculated using the STAND X FIRE interaction term as the denominator (Littell and others 1993). Furthermore, the F-statistic for all within plot factors were calculated using the residual mean squared error as the denominator.

In a separate analysis, red maple, sugar maple, American beech, and basswood each occurred in enough plots to test if there was differential survival among species. Species had to occur in at least four plots in both size classes to be included in the analysis. For these species level analyses, we conducted a split-plot ANOVA as above, except with SPECIES (instead of NO-DEER) as a within plot factor. We used least squared means and orthogonal contrasts to examine significant interactions.

Sapling Sprouting Response

We calculated the percentage of top-killed saplings (1-10 cm DBH) that sprouted for the 14 most common species per plot. Because percent sprouting for many species and plots was close to 0 percent, we performed an arcsine transformation on all of the proportional sprouting data to normalize the distributions (Sokal and Rohlf 1995). We classified top-killed saplings with sprouts into two overlapping categories:

(1) all individuals with sprouts greater than 1 cm tall and
(2) all individuals with sprouts greater than 50 cm tall.

We divided sprouts into these two categories because this identified vigorous sprouts from the ones that sprouted and failed to grow. We used an ANOVA with a split-plot, two-way factorial design to test for the effects of the treatments (FIRE, NO-DEER, GAPS) and their interactions on mean sprouting response for all species combined that had sprouts that were greater than 1 cm tall. FIRE was the main factor and NO-DEER, GAP and NO-DEER X GAP were the within plot factors.

We conducted three analyses on saplings with tall sprouts (> 50 cm tall). First, we used an ANOVA with a split-plot, two-way factorial design to test for the effects of the treatments (FIRE, NO-DEER, GAP) and their interactions on mean sprouting response for all species combined. FIRE was the main factor and NO-DEER, GAP, and NO-DEER X GAP were the within plot factors. Second, we tested for the effects of FIRE, GAP, and SPECIES on sprouting response in the absence of herbivory (i.e., all of the NO-DEER plots: NO-DEER, NO-DEER X GAP, FIRE X NO-DEER, and FIRE X NO-DEER X GAPS). We used an ANOVA with the same split-plot, two-way factorial design as the previous analysis except with SPECIES, instead of NO-DEER, as a within plot factor. Here a significant interaction of SPECIES X FIRE or SPECIES X GAP demonstrates species-specific sprouting responses to fire or gaps in the absence of any browsing pressure. The species we included in this analysis were striped maple, red maple, sugar maple, American beech and black cherry because each occurred in at least three plots.

Lastly, we further examined the effects of gaps and species on sprouting response on 14 species in the absence of herbivory and following fire (i.e., FIRE X NO-DEER plots versus FIRE X NO-DEER X GAP PLOTS). We used a two-way factorial ANOVA with SPECIES, GAP, and

their interaction as factors. If the GAP X SPECIES interaction was not significant, we combined these two treatments and compared sprouting response of 14 species in the community. The species included were: striped maple, red maple, sugar maple, birch, hickory, American beech, white ash, witch hazel, yellow poplar, magnolia, ironwood, black cherry, northern red oak, and basswood. We used least squared means and orthogonal contrasts to explore significant interactions.

RESULTS

Canopy Tree Survival

As expected, deer browsing did not significantly influence the survival of canopy trees ($P > 0.2$; table 2). Therefore, for all remaining canopy survival analyses, we pooled control plots and no-deer plots (the no fire treatment) and pooled fire plots and fire x no-deer plots (the fire treatment). For all species combined, fire killed a significant proportion of small canopy trees

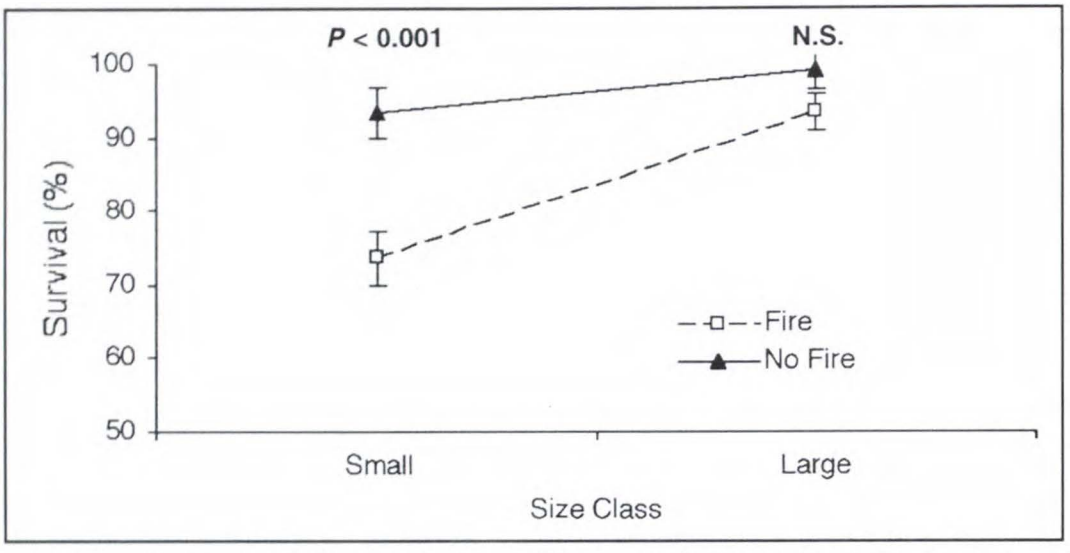

Figure 1.—*Mean canopy tree survival (± 1 SE) of trees in fire and no-fire treatments and in two size classes: large (> 20 cm DBH) and small (10-19.9 DBH).*

Table 2.—*Mean canopy tree survival (± 1 SE) per 400 m² plot for 10 common species and two size classes*

	Fire treatments[1]						No-fire treatments[2]					
	Small 10-19.9 cm DBH			Large ≥20 cm DBH			Small 10-19.9 cm DBH			Large ≥20 cm DBH		
	Mean	SE	N[3]	Mean	SE	N[3]	Mean	SE	N	Mean	SE	N[3]
ACRU[4]	50.00	22.36	6	80.95	14.29	7	100.00	0.00	8	100.00	0.00	10
ACSA	75.48	7.51	15	100.00	0.00	14	95.40	2.71	14	100.00	0.00	13
BESP	100.00		1	100.00	0.00	4	100.00	0.00	2	94.44	5.56	9
FAGR	78.57	16.39	6	85.71	14.29	7	100.00	0.00	6	96.30	3.70	9
FRAM	50.00		1	100.00	0.00	3			0	100.00	0.00	6
LITU	100.00		1	92.86	7.14	7	100.00		1	100.00	0.00	5
MASP	100.00		1	70.00	20.00	5	80.00	20.00	5	100.00	0.00	4
PRSE			0	95.83	4.17	8			0	100.00	0.00	6
QURU			0	100.00	0.00	7			0	100.00	0.00	11
TIAM	80.00	20.00	5	100.00	0.00	9	75.00	25.00	4	100.00	0.00	6

[1] Fire treatments included fire and fire x no-deer plots (Maximum N = 16).
[2] No-fire treatments included control and no-deer plots (Maximum N = 16).
[3] N varies because not all species and size classes occurred in all plots.
[4]Species names are described Appendix table 1.

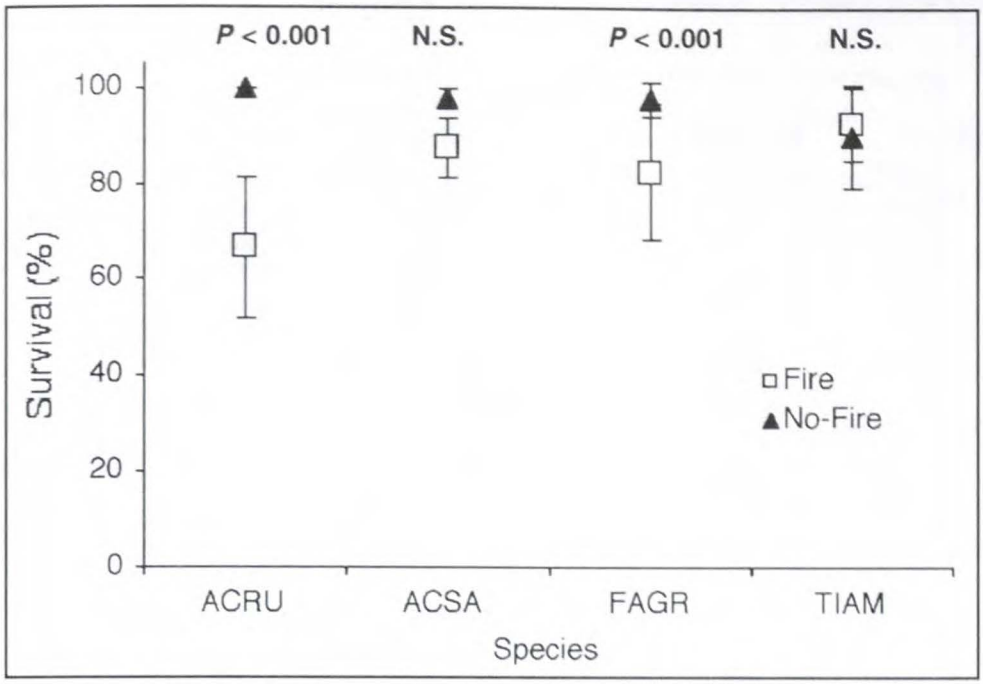

Figure 2.—*Mean canopy tree survival (± 1 SE) of trees in fire and no-fire treatment for red maple, sugar maple, American beech, and basswood.*

(10-19.9 cm DBH) but had no significant effect on large canopy trees (> 20 cm DBH; $p < 0.05$, table 2, fig. 1). We found no significant effect of fire on large canopy trees.

There were no species-specific effects of size regardless of fire (FIRE X SIZE X SPECIES interaction not significant). Fire killed a greater proportion of primarily small red maple and American beech (fig. 2), and had no significant effect on sugar maple and basswood (FIRE X SPECIES interaction, $p < 0.05$, Appendix table 3). Note that there were no individuals of oak in the small canopy size classes (table 2).

Sprouting Behavior

General Sprouting Responses

Of the nearly 8,000 individual saplings we monitored, 4,155 were classified as top-killed or top-killed with sprouts during the second growing season following fire. A greater proportion of top-killed saplings produced sprouts (>1 cm tall) in fire plots and in no-deer plots than saplings without fire or browsed by deer (fig. 3A). Additionally, top-killed saplings of most species produced sprouts (table 3). Gaps, however, did not significantly affect the proportion of top-killed saplings that sprouted ($P > 0.6$; table 4). We restricted the remaining analyses to top-killed saplings that produced tall, vigorous sprouts greater than 50 cm tall.

Tall Sprouts In Fire And No Fire Areas

Deer browsing decreased the number of top-killed saplings that produced tall sprouts (fig. 3B). This response was greatly magnified in the fire treatment (fig. 4). Following fire, deer browsing reduced diversity of large sprouts from 12 to 2 species (table 4). Only striped maple and red maple had large sprouts outside of fences in fire plots. Gaps did not have a significant effect on sprouting behavior ($P > 0.1$; table 5).

Tall Sprouts In The No-Deer Plots Including Fire And No Fire Areas

We tested for interspecific differences in sprouting behavior among the five most abundant species that had top-killed individuals in both the fire and no fire areas. We used only five species to compare fire and no fire areas because so few individuals were top-killed in the unburned areas. Additionally, we only conducted this analysis for no-deer plots because browsing by deer drastically reduced the number of species that produced sprouts over 50 cm (table 4). Our goal was to evaluate species-specific sprouting potential without the pervasive and damaging effect of browsing. Again, gap did not affect sprouting response ($P > 0.3$, Appendix table 6). Figure 5 shows that fire did have a species-specific effect on sapling sprouting (FIRE X SPECIES interaction, $P < 0.001$). Striped maple and black cherry sprouted in similar proportions in the burned and unburned plots ($P > 0.05$). Top-killed red maple

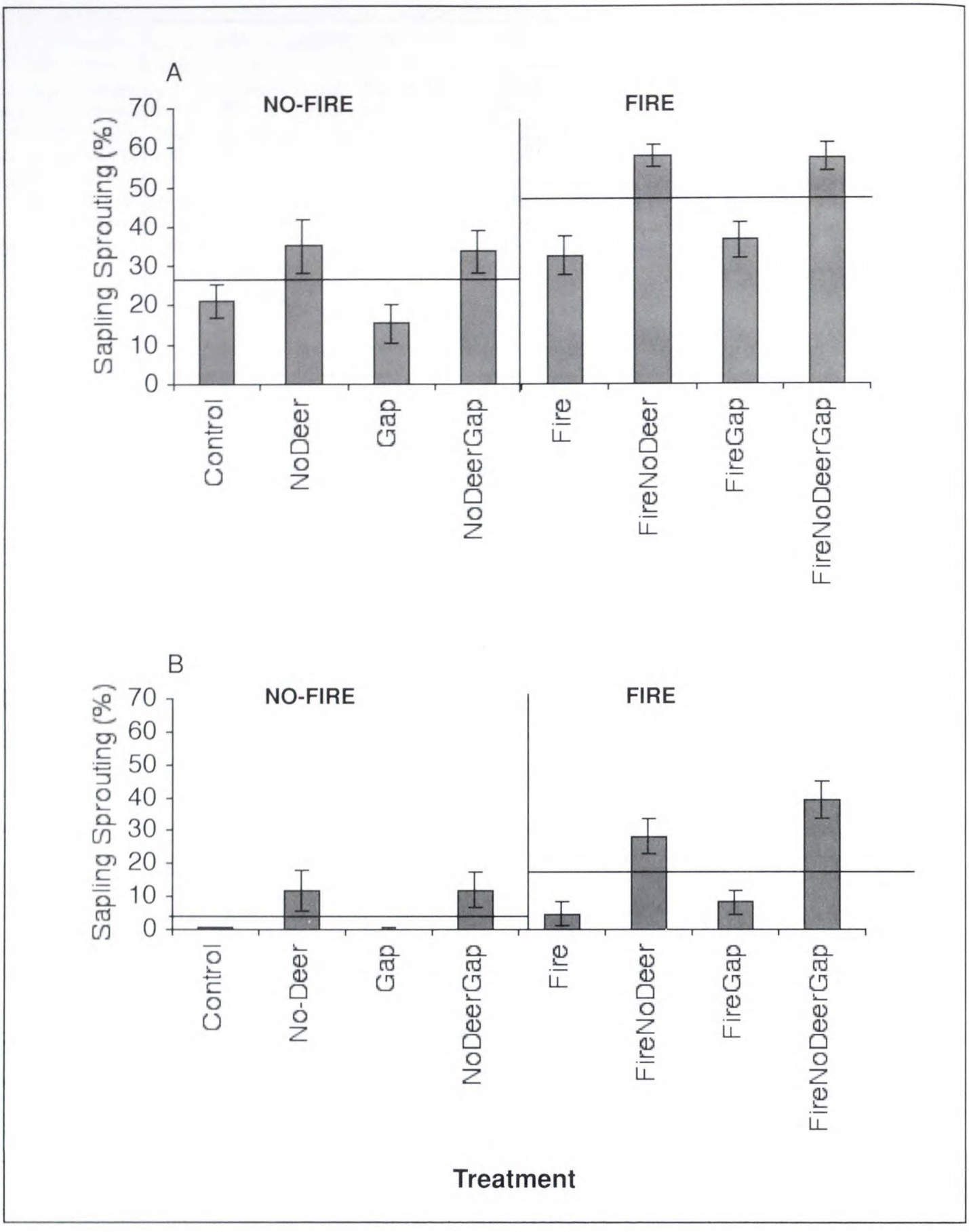

Figure 3.—*Mean percent sprouting of top-killed saplings for all species combined within each treatment (± 1 SE; N=8) for (A) all sprouts (> 1 cm tall) and for (B) tall sprouts (> 50 cm tall). All species are combined. Horizontal lines are fire and no-fire treatment means.*

Table 3.—*Mean percent (± 1 SE) of top-killed saplings that produced sprouts > 1 cm tall in eight treatments. Means > 5% in > 1 plot are in bold.*

	NO-FIRE TREATMENTS											
	Control			No-Deer			Gap			No-deer x gap		
Species[1]	Mean	SE	N[2]	Mean	SE	N[2]	Mean	SE	N[2]	Mean	SE	N[2]
ACPE	**36.02**	**11.75**	**6**	**29.72**	**15.09**	**6**	4.76	4.76	3	**55.00**	**20.00**	**5**
ACRU	0.00	0.00	4	25.00	25.00	2	**11.11**	**11.11**	**3**	100.00	.	1
ACSA	4.28	2.28	7	**31.93**	**10.70**	**7**	3.31	2.36	8	**29.29**	**13.65**	**7**
BESP	0.00	0.00	4	0.00	0.00	2	0.00	0.00	2	0.00	0.00	2
CASP	**75.00**	**25.00**	**2**	100.00	.	1				0.00	.	1
FAGR	**37.50**	**21.04**	**4**	**24.67**	**10.41**	**5**	**20.83**	**12.50**	**4**	**33.33**	**33.33**	**3**
FRAM	60.00	.	1				**8.33**	**8.33**	**3**	100.00	.	1
HAVI										0.00	.	1
LITU							100.00	.	1			
MASP	**16.67**	**16.67**	**3**	100.00	.	1	**16.67**	**16.67**	**2**	100.00	.	1
OSVI	0.00	0.00	3				0.00	0.00	2			
PRSE	**47.50**	**16.01**	**5**	**56.82**	**16.89**	**3**	**56.57**	**21.73**	**3**	**45.00**	**5.00**	**2**
QURU	33.33	.	1				**50.00**	**50.00**	**2**			
TIAM	50.00	.	1	0.00	0.00	2	0.00	.	1	0.00	.	1

	FIRE TREATMENTS											
	Fire			Fire x no-deer			Fire x gap			Fire x no-deer x gap		
Species	Mean	SE	N[2]	Mean	SE	N[2]	Mean	SE	N[2]	Mean	SE	N[2]
ACPE	**24.01**	**9.47**	**7**	**43.19**	**8.52**	**8**	**55.03**	**13.75**	**6**	**56.03**	**8.33**	**5**
ACRU	**55.56**	**29.40**	**3**	**91.67**	**8.33**	**4**	**44.74**	**22.37**	**3**	**63.05**	**12.71**	**5**
ACSA	**27.89**	**5.52**	**8**	**60.53**	**2.39**	**8**	**16.18**	**4.62**	**8**	**60.06**	**5.68**	**8**
BESP	0.00	0.00	3	**14.06**	**12.07**	**4**	0.00	0.00	5	0.00	0.00	5
CASP	66.67	.	1	**66.67**	**33.33**	**3**	0.00	.	1	100.00	.	1
FAGR	**72.22**	**8.81**	**5**	**53.92**	**12.76**	**5**	**56.64**	**14.06**	**6**	**53.63**	**13.92**	**7**
FRAM	0.00	0.00	2	0.00	.	1	**33.33**	**33.33**	**2**	0.00	.	1
HAVI	**42.75**	**9.42**	**2**	**59.62**	**9.62**	**2**	50.00	.	1	**66.67**	**9.62**	**3**
LITU	**25.00**	**25.00**	**2**	**68.75**	**23.66**	**4**	**50.00**	**50.00**	**2**	**50.00**	.	**1**
MASP	**33.33**	**33.33**	**3**	50.00	.	1	**16.67**	**16.67**	**3**	**66.67**	**33.33**	**3**
OSVI	**16.67**	**16.67**	**3**	**23.47**	**2.41**	**3**	**61.11**	**20.03**	**3**	**13.33**	**13.33**	**3**
PRSE	**36.15**	**11.11**	**5**	**76.07**	**8.30**	**5**	**48.84**	**17.80**	**5**	**64.04**	**19.11**	**4**
QURU	0.00	.	1	0.00	.	1	**50.00**	**50.00**	**2**	0.00	0.00	3
TIAM	0.00	.	1	**25.00**	**25.00**	**2**	**46.67**	**22.61**	**5**	**33.33**	**33.33**	**3**

[1] Maximum possible N is 8. N varies because not all species occurred in all plots.
[2] Species names are described in Appendix table 1.

and sugar maple sprouted copiously following fire, but did not sprout in unburned areas ($P < 0.01$).

Tall Sprouts In No-Deer Plots In Fire Plots Only

We examined the interspecific differences in sprouting behavior among the 14 most abundant species within burned sites. GAP and GAP X SPECIES interaction did not affect sprouting response ($P > 0.15$ and $P > 0.76$, respectively). Therefore, we combined data from FIRE X NO-DEER plots with FIRE X NO-DEER X GAP plots to evaluate species-specific sprouting responses (fig. 6). Hickory, yellow poplar, witch hazel, magnolia, red maple, and black cherry all sprouted in greater proportion than the mean for all individuals (31 percent) for all the common species in the community. Striped maple, basswood, and sugar maple had an intermediate response at or near the mean. Ironwood, birch, American beech, white ash, and northern red oak all sprouted at proportions well below the mean.

DISCUSSION

We examined the fire and oak hypothesis in mixed hardwood stands where oaks were present but uncommon. Others have suggested that mixed hardwood stands with a low occurrence of oak regeneration best characterize the classic failed oak regeneration pattern due to fire suppression (Abrams 1992, Lorimer 1992, Brose

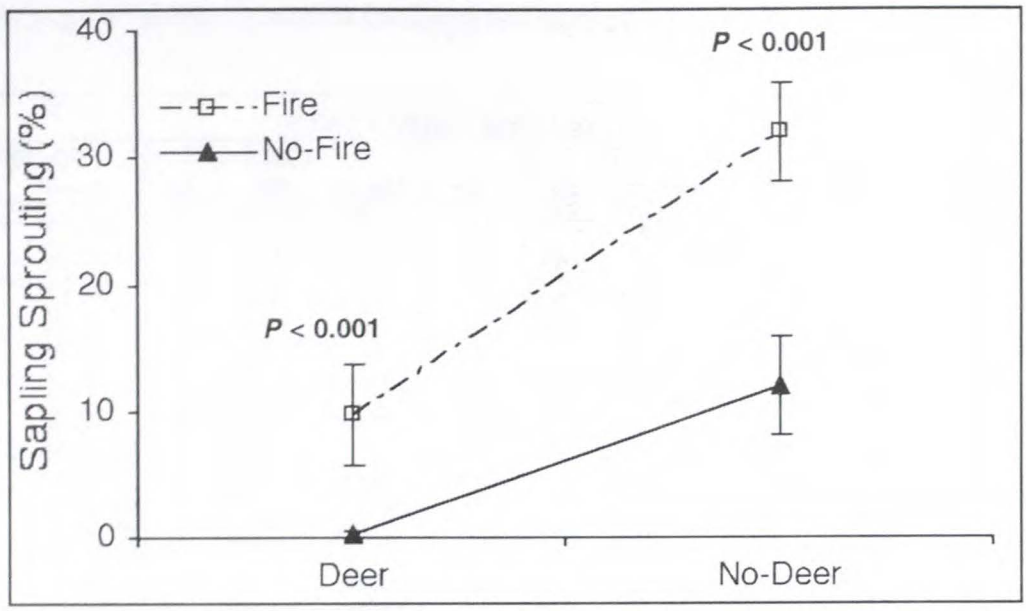

Figure 4.—*Mean percent sprouting (± SE) of top-killed saplings with sprouts > 50 cm tall in fire and no-fire treatments and deer and no-deer treatments.*

and others 2001). When prescribed fires have failed to increase the relative abundance of oak, others have suggested that fires were too low in intensity or a single fire is insufficient to promote oak abundance. Published evidence that multiple fires promote oak abundance is weak. Nyland and others (1982) had one site where they conducted two fires and presented no vegetation data or statistics. Merritt and Pope (1991) found that the effects of two fires on northern red oak seedlings were not different from the effects of one fire. Lastly, Huddle and Pallardy (1996) found that multiple fires decreased red oak abundance. The areas we burned in this study likely had historical fire return times on the century scale and did not experience frequent understory fires set by pre-Columbian humans as occurred in other regions (Abrams 1992, Guyette and Dey 2000, Brose and others 2001).

Fire could promote oak's dominance in forest communities in two ways: by promoting the relative density of oak while leaving other species unchanged or by decreasing the relative density of co-occurring species and leaving oaks unchanged or a combination of both of these. Lorimer (1992) suggested that the latter is more important but, to date, empirical evidence is equivocal. Determining which of these two processes is more prevalent is important to understanding how fire and its interactions with other processes structure communities. In this study, we examined the relative contribution of these two processes.

Testing Prediction 1: Canopy Tree Survival

The first prediction we tested was that northern red oak canopy trees will survive fire in a higher proportion than co-occurring species. Even though fire caused species-specific mortality in small canopy trees, we could not test this prediction because there were no small oak canopy trees. Large northern red oak canopy trees (> 20 cm DBH) had 100 percent survival following fire, but so did large sugar maple, white ash, basswood, and birch. The latter two species are considered extremely fire intolerant (Burns and Honkala 1990). Although we did not find support for the fire and oak hypothesis in canopy trees, our data do demonstrate that even "fire intolerant" species can reach size refuges.

Size refuges, above which fire can not extirpate canopy tree species from an area, have been documented for some species since at least the mid-1980s (White 1983, Harmon 1984). In a 3-decade long study, Huddle and Pallardy (1996) demonstrated that post oak (*Q. stellata*) reached a size refuge around 15 to 20 cm DBH, above which repeated fires did not affect survival. Hickory species could also reach a size refuge under some conditions. Conversely, red oak species, which included northern red oaks, never reached a size refuge. Survival was based on canopy pretreatment size and the authors did not report sizes greater than 20 cm DBH (Huddle and Pallardy 1996).

In two other elaborate and extensive prescribed fire experiments, however, researchers failed to take canopy tree size into account in analyses

Table 4.—*Mean percent (± 1 SE) of top-killed saplings that produced sprouts > 50 cm tall in eight treatments. Means > 5% in > 1 plot are in bold*

NO-FIRE TREATMENTS												
	Control			No-deer			Gap			No-deer x gap		
Species[1]	Mean	SE	N[2]	Mean	SE	N[2]	Mean	SE	N[2]	Mean	SE	N[2]
ACPE	0.00	0.00	6	**18.75**	**16.38**	**6**	1.59	1.59	3	**42.50**	**23.58**	**5**
ACRU	0.00	0.00	4	0.00	0.00	2	0.00	0.00	3	0.00	.	5
ACSA	0.00	0.00	7	2.86	2.86	7	0.00	0.00	8	2.86	2.86	8
BESP	0.00	0.00	4	0.00	0.00	2	0.00	0.00	2	0.00	0.00	5
CASP	0.00	0.00	2	0.00	.	1	.	.	0	0.00	.	1
FAGR	0.00	0.00	4	0.00	0.00	5	0.00	0.00	4	**33.33**	**33.33**	**7**
FRAM	0.00	.	1	.	.	0	0.00	0.00	3	0.00	.	1
HAVI	.	.	0	.	.	0	.	.	0	0.00	.	3
LITU	.	.	0	.	.	0	0.00	.	1	.	.	1
MASP	0.00	0.00	3	100.00	.	1	0.00	0.00	2	**100.00**	.	**3**
OSVI	0.00	0.00	3	.	.	0	0.00	0.00	2	.	.	3
PRSE	1.00	1.00	5	**23.78**	**16.53**	**3**	0.00	0.00	3	**45.00**	**5.00**	**4**
QURU	0.00	.	1	.	.	0	0.00	0.00	2	.	.	3
TIAM	0.00	.	1	0.00	0.00	2	0.00	.	1	0.00	.	3

FIRE TREATMENTS												
	Fire			Fire x no-deer			Fire x gap			Fire x no-deer x gap		
Species	Mean	SE	N[2]	Mean	SE	N[2]	Mean	SE	N[2]	Mean	SE	N[2]
ACPE	**12.23**	**7.12**	**7**	**27.99**	**9.41**	**8**	**27.57**	**10.98**	**6**	**43.53**	**9.29**	**5**
ACRU	**11.11**	**11.1**	**3**	**58.33**	**25.00**	**4**	0.00	0.00	3	**54.38**	**13.57**	**5**
ACSA	0.09	0.09	8	**21.19**	**2.80**	**8**	0.45	0.45	8	**37.22**	**6.60**	**8**
BESP	0.00	0.00	3	**12.50**	**12.50**	**4**	0.00	0.00	5	0.00	0.00	5
CASP	0.00	.	1	**55.56**	**29.40**	**3**	0.00	.	1	100.00	.	1
FAGR	0.00	0.00	5	0.00	0.00	5	0.62	0.62	6	**7.14**	**7.14**	**7**
FRAM	0.00	0.00	2	0.00	.	1	0.00	0.00	2	0.00	.	1
HAVI	0.00	0.00	2	**59.62**	**9.62**	**2**	0.00	.	1	**66.67**	**9.62**	**3**
LITU	0.00	0.00	2	**68.75**	**23.66**	**4**	0.00	0.00	2	50.00	.	1
MASP	0.00	0.00	3	50.00	.	1	0.00	0.00	3	**66.67**	**33.33**	**3**
OSVI	0.00	0.00	3	**12.92**	**6.47**	**3**	0.00	0.00	3	**13.33**	**13.33**	**3**
PRSE	0.00	0.00	5	**51.10**	**14.23**	**5**	0.53	0.53	5	**44.41**	**21.42**	**4**
QURU	0.00	.	1	0.00	.	1	0.00	0.00	2	0.00	0.00	3
TIAM	0.00	.	1	**25.00**	**25.00**	**2**	0.00	0.00	5	**33.33**	**33.33**	**3**

[1] Species are described in Appendix table 1.
[2] Maximum possible N is 8. N varies because not all species occurred in all plots.

(Elliot and others 1999, Brose and Van Lear 1999). Elliot and others (1999) suggested that:

1) red maple (> 5 cm DBH) density decreased at high, fire intensity,
2) did not change at low fire intensity, and
3) that northern red oak (> 5 cm DBH) densities did not change across fire intensities.

In that study, it is possible that maples on average were smaller than the oaks. Similarly, Brose and Van Lear (1999) reported that oaks had about an 80 percent survival following fire compared to American beech and red maples that combined had about 50 percent survival after spring burns. However, they could not disentangle the effects of size from the effects of species because the oaks were overstory trees and the American beech and red maples were midstory trees (Brose and Van Lear 1999). Here, we reported survival of canopy trees in the second growing season following fire. Fire may have differentially damaged some trees that may die in the future causing the long term survival to differ from these short term results. However, we compared our results with other studies that reported survival of canopy trees 2 to 3 years following fire (Brose and Van Lear 1999, Elliot and others 1999.

The increase in abundance of red maple is considered a broad, region-wide, management

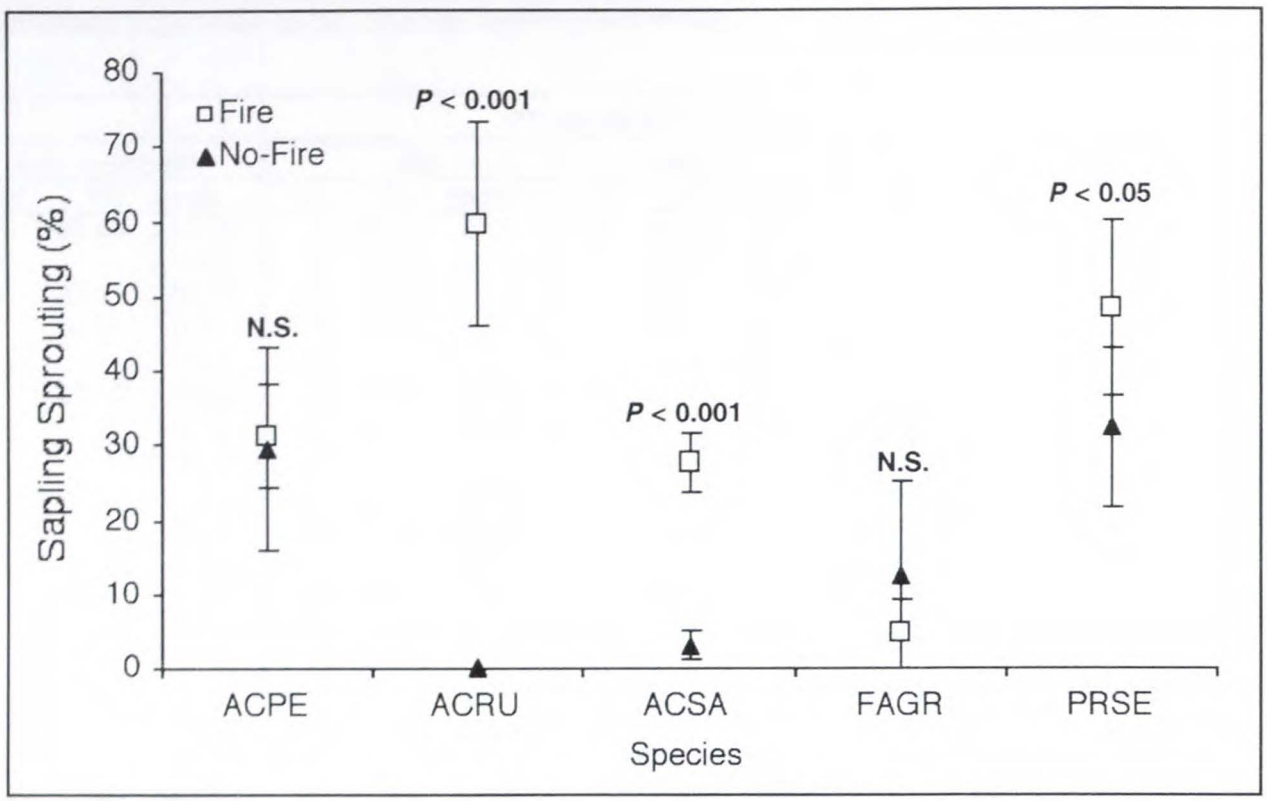

Figure 5—*Mean percent sprouting (± SE) of top-killed saplings with sprouts > 50 cm tall in fire and no-fire treatments for five common species: striped maple, red maple, sugar maple, American beech, and basswood.*

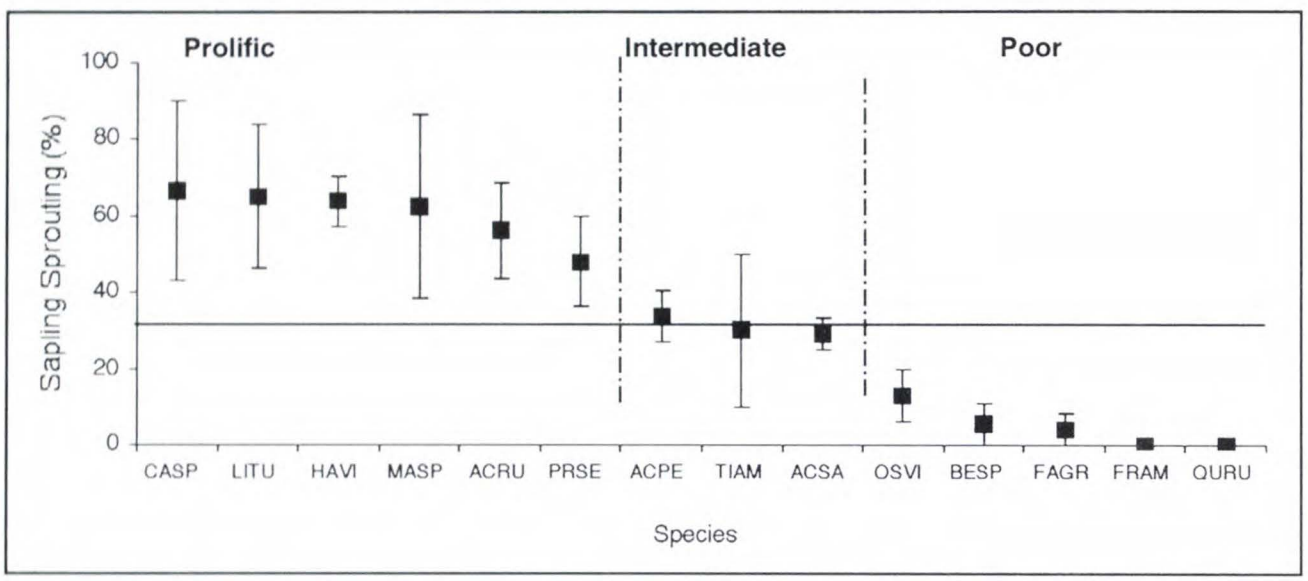

Figure 6.—*Mean sprouting of top-killed saplings (sprouts > 50 cm tall) ranked in descending order for 14 species in fire treatments (fire x no-deer plots and fire x no-deer x gap plots). The horizontal lines are the mean sprouting responses for the community. Species names are listed in Appendix table 1.*

problem (Abrams 1998). Advocates of the fire and oak hypothesis champion fire as a management tool that could reduce red maple while increasing northern red oak abundance (Van Lear and Watt 1992, Brose and others 2001). Our results suggest that fire can kill a significant proportion of small canopy red maples while leaving large northern red oak unaffected. However, no studies to date have adequately demonstrated comparative size-specific survival for both species. Such information is critical for managers. Additionally, Huddle and Pallardy (1996) recommend that managers should consider species composition, stand age, and other environmental stressors before applying prescribed fire, we concur.

Testing Prediction 2. Sapling Sprouting

The second predictions we tested was that top-killed northern red oak saplings would sprout in higher proportions than co-occurring species following fire. Oak saplings did not produce tall, vigorous sprouts that were likely to persist during this study. Many authors have argued that sprouting is the mechanism by which oak seedlings and saplings are fire tolerant (Crow 1988, Abrams 1992, Van Lear and Watt 1992, Johnson 1993, Brose and Van Lear 1998, Huddle and Pallardy 1999). However, evidence from the literature is ambiguous. Kruger and Reich (1997) conducted small burns (~200 m^2) in 0.5 ha openings that were fenced to remove deer browsing. Because nearly all seedlings were top-killed, the post-burn densities represented mainly seedling sprouting (~20 to 50 cm tall pre-treatment size). They found that northern red oak increased in relative density by nearly 50 percent whereas sugar maple density decreased by nearly 90 percent. Black cherry densities were unchanged.

In another study, Huddle and Pallardy (1999) planted nursery stock in a field that was then burned. Northern red oak survival was nearly four times greater than red maple survival. They attributed most of the oak seedling survival to sprouting. Others have demonstrated that northern red oak had superior sprouting compared to red maple, sugar maple, American beech, and white ash (Christianson 1971, Swan 1970). Johnson (1974) reported that 40 percent of naturally growing, 2-year-old northern red oak seedlings sprouted following fire. Limitations in these previous studies included:

1) sprouting often could not be separated from survival results, or
2) sprouting was inferred from density or survival results, or
3) studies focused on seedlings (i.e., < 140 cm tall; Christianson 1971, Johnson 1974, Kruger and Reich 1997, Huddle and Pallardy 1999, Brose and Van Lear 1998, Elliott and others 1999).

Saplings (1-10 cm DBH) are more likely than seedlings to replace canopy trees in gaps and after stand removal disturbances for some species (Sander 1971, Loftis 1983, Runkle 1990; but see Schuler and Fajvan 1999).

There is no question that understory northern red oaks can be fire tolerant via sprouting in some conditions. However, we found that top-killed northern red oak saplings did not produce tall, vigorous sprouts. This result demonstrates that northern red oak is not a ubiquitous sprouter in all conditions. Northern red oak's fire tolerance may vary across gradients of site characteristics, forest types, or fire conditions which may explain why this species is fire tolerant in some experiments, but not in others. Our result that northern red oak sprouted in lower proportions than red maple does concur with results from some other studies. Arthur and others (1998) found that after spring fires, red maple sprouts were 3 times more abundant than chestnut oak sprouts and 10 times more abundant than scarlet oak sprouts. Wendel and Smith (1986) reported that red maple sprouted prolifically and northern red oak sprouted poorly following fire. Through a different approach, Clark and his colleagues (Clark and others 1996, Clark and Royall 1996, Clark 1997) cast doubt on the ubiquitous application of the fire and oak hypothesis (but see Abrams and Seischab (1997) for a reply). They demonstrated through fossil pollen and charcoal records that oaks have been prevalent in areas and times with fire intolerant species (i.e., American beech) and during periods of low fire frequency (Clark and others 1996, Clark and Royall 1996). Our work, and that of Clark and his colleagues, suggests that fire alone does not explain the occurrence of oaks, and in particular northern red oak, in some areas.

Incorporating Deer and Gaps Into The Fire and Oak Hypothesis

We found that deer browsing, after prescribed fires, dramatically reduced the proportion of all top-killed saplings that sprouted, reduced diversity of sprouts by an order of magnitude, and greatly altered species composition of sprouters. Only striped maple and red maple produced large sprouts following fire in the presence of deer browsing. Because sprouting is a major mechanism of regeneration in these forests,

deer browsing has the potential to alter successional pathways following fire. In fact, in the absence of deer herbivory, striped maple was an intermediate sprouter but dominated the sprouting community in the presence of herbivory. These interactions between fire and deer demonstrate that fire alone does not always control species composition.

Red maple, sugar maple, and witch hazel saplings have been reported to be fire intolerant and have a low occurrence of sprouting following fire (Wright and Bailey 1982, Burns and Honkala 1990, Huddle and Pallardy 1999). However, we found that all three of these species sprouted at or above the mean sprouting response for the community. Red maple saplings were very fire tolerant and sprouted prolifically even in the presence of deer herbivory. This suggests that the increase in red maple abundance in recent decades may be due not to fire suppression but due to red maple's ability to tolerate deer herbivory. Witch hazel was a prolific sprouter, and sugar maple was an intermediate sprouter both in the absence of deer herbivory. Sugar maple is likely to maintain its dominance following fire in the no-deer plots because it was the dominant sapling in the pretreatment community. Previous work on sugar maple and witch hazel attributed poor sprouting post fire to intolerance to fire (Wright and Bailey 1982, Burns and Honkala 1990). Our results suggest that these two species are tolerant to fire but intolerant to deer browsing.

Unlike browsing and fire, gaps did not alter the proportion of top-killed saplings that sprouted. This was surprising considering the number of published studies on the importance of gaps for many different modes of regeneration including seed germination, seedling and sapling growth, the role of epiphytes, and effects of vines (Canham 1985, Lorimer and others 1988, Runkle 1990, Spies and others 1990, Lawton and Putz 1988, Clebsch and Busing 1989, Schnitzer and others 2000). However, the role of gaps in sprouting is largely unstudied. We found that the high-resource levels in gaps did not alter the proportion of top-killed saplings that sprouted for 14 common species in 32 multi-tree gaps. Further, the effects of gaps did not interact with the effects of fire or deer browsing. Lastly, there were no significant species-specific effects of gaps on sprouting response regardless of fire treatment. This study demonstrates that gaps play little role in the initial sprouting responses of top-killed saplings in this system.

IMPLICATIONS FOR MANAGEMENT

In a recent opinion paper, Brose and others (2001) advocated for a wider use of prescribed fire for the management of oak in mixed hardwood stands. Our results demonstrate that prescribed fire can be very detrimental to sprouting of northern red oak saplings in mixed hardwood stands. Furthermore, prescribed fire in the presence of large deer populations prevented all commercial species from producing vigorous, tall sprouts, except red maple. For decades, deer populations have been at historically high levels in many areas in eastern North America and have been adversely impacting understory vegetation (Beals and others 1960, Marquis 1981, Alverson and others 1988, Tilghman 1989). This suggests that our results on the effects of deer browsing may have broad implications. Undoubtedly, some fires do promote oak regeneration especially in some oak dominated communities; however, fires do not always promote northern red oak in mixed hardwood stands. Additional research testing the effects of fire on other eastern deciduous forest types in the context of deer browsing and other factors will likely elucidate mechanisms by which the fire and oak hypothesis operates.

CONCLUSIONS

Our findings only partially support the fire and oak hypothesis. Canopy northern red oaks were fire tolerant (i.e., had 100 percent survival), however, oak saplings were not fire tolerant and did not produce tall, vigorous sprouts following fire. Further, our findings demonstrate that deer browsing following fire can dramatically reduce regeneration via sprouting leading to communities dominated by striped maple. Although some of these specific results may not be generalizable to all regions within the eastern deciduous forest region, they do cast doubt on the ubiquitous application of the fire and oak hypothesis to explain the dominance of oak in some areas. These findings illustrate that prescribed fires in the presence of high deer populations may prevent forest regeneration and likely shift successional trajectories.

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Appendix Table 1.—*Codes, genus species and authority according to Gleason and Cronquist (1991), and common name for plant species found on the study sites and discussed in the paper*

Species Code	Scientific name	Common name
ACPE	*Acer pensylvanicum* L.	striped maple
ACRU	*Acer rubrum* L.	red maple
ACSA	*Acer sacharrum* Marsh.	sugar maple
BESP	*Betula lenta* L.	birch
BESP	*Betula alleghaniensis* Britton	birch
CASP	*Carya tomentosa* Nutt.	hickory
CASP	*Carya ovata* (Mill.) K. Koch	hickory
FAGR	*Fagus grandifolia* Ehrh.	American beech
FRAM	*Fraxinus americana* L.	white ash
HAVI	*Hamamelis virginiana* L.	witch hazel
LITU	*Liriodendron tulipifera* L.	yellow poplar
MASP	*Magnolia acuminata* L.	magnolia
MASP	*Magnolia fraseri* Walt.	magnolia
OSVI	*Ostrya virginiana* (Mill.) K. Koch	ironwood
OXAR	*Oxydendrum arboreum* (L.) DC.	sourwood
PRSE	*Prunus serotina* Ehrh.	black cherry
QUPR	*Quercus prinus* L.	chestnut oak
QURU	*Quercus rubra* L.	northern red oak
ROPS	*Robinia pseudoacacia* L.	black locust
TIAM	*Tilia americana* L.	basswood
TSCA	*Tsuga canadensis* (L.) Carr.	eastern hemlock

Appendix Table 2.—*Results of a ANOVA with a split-plot, two-way factorial design testing the effects of Fire, No-Deer and Size on canopy tree survival*

Source	df	SS	MS	*F*	*P*	R^2
MODEL	13	3.0373	0.2336	4.61	0.0001	0.54
ERROR	50	2.53120	0.0506			

Source	df	Type III SS	Type III MS	*F*	*P*
STAND	3	0.6902			
STAND x FIRE	3	0.4901	0.1634		
FIRE	1	0.7653	0.7653	4.68	0.1191
NO-DEER	1	0.0800	0.0800	1.59	0.2130
SIZE	1	0.6359	0.6359	12.64	0.0008
FIRE x NO-DEER	1	0.0251	0.0251	0.50	0.4833
FIRE x SIZE	1	0.2531	0.2531	5.03	0.0292
NO-DEER x SIZE	1	0.0638	0.0638	1.27	0.2652
FIRE x NO-DEER x SIZE	1	0.0336	0.0336	0.336	0.4188

Appendix Table 3.—*Results of an ANOVA with a split-plot, two-way factorial design testing the effects of Fire, Size, and Species on canopy tree survival*

Source	df	SS	MS	*F*	*P*	R^2
Model	21	11.9006	0.5667	5.980	0.0001	0.52
Error	117	11.0888	0.09478			

Source	df	Type III SS	Type III MS	*F*	*P*
Stand	3	2.7215			
Stand x Fire	3	2.2731	0.7577		
Fire	1	1.9076	1.9076	2.52	0.2108
Species [1]	3	0.6667	0.6667	2.34	0.0765
Size[2]	1	1.9379	1.9379	20.45	0.0001
Fire x Size	1	0.5267	0.5267	5.56	0.0201
Fire x Species	3	1.2339	1.2339	4.34	0.0061
Size x Species	3	0.4865	0.4865	1.71	0.1685
Fire x Size x Species	3	0.2691	0.2691	0.95	0.4206

[1] The four species were red maple, sugar maple, American beech, and basswood.
[2] Size class for trees are Small: 10-19.9 cm DBH; and Large: ≥ 20 cm DBH.

Appendix Table 4.—*Results of an ANOVA with a split-plot, two-way factorial design testing the effects of FIRE, GAP, NO-DEER and their interactions on mean proportion of top-killed sapling with sprouts > 1 cm tall*

Source	df	SS	*F*	*P*	R^2
Model	13	1.8262	4.14	0.0001	0.52
Error	50	1.6974			

Source	df	Type III SS	Type III MS	*F*	*P*
Stand	3	0.0377			
Stand x Fire	3	0.8954	0.0298		
Fire	1	0.9643	0.9643	32.31	0.0108
Gap	1	0.0086	0.0086	0.25	0.6158
No-Deer	1	0.7078	0.7078	20.85	0.0001
Fire x Gap	1	0.0513	0.0513	1.51	0.2245
Fire x No-Deer	1	0.0017	0.0017	0.05	0.8227
Gap x No-Deer	1	0.0000	0.0000	0.00	0.9863
Fire x Gap x No-Deer	1	0.0285	0.0285	0.84	0.3636

Appendix Table 5.—*Results of an ANOVA with a split-plot, two-way factorial design testing the effects of FIRE, GAP, NO-DEER and their interactions on mean proportion of top-killed sapling with tall sprouts (> 50 cm tall)*

Source	df	SS	*F*	*P*	R^2
MODEL	13	3.7702	9.36	0.0001	0.71
ERROR	50	1.5495			

Source	df	Type III SS	Type III MS	*F*	*P*
STAND	3	0.9751			
STAND x FIRE	3	0.0503	0.0168		
FIRE	1	1.1822	1.1822	70.47	0.0035
GAP	1	0.0681	0.0681	2.20	0.1444
NO-DEER	1	1.6454	1.6454	53.10	0.0001
FIRE x GAP	1	0.0640	0.0640	2.06	0.1571
FIRE x NO-DEER	1	0.1350	0.1350	4.35	0.0420
GAP X NO-DEER	1	0.0001	0.0001	0.00	0.9518
FIRE x NO-DEER x GAP	1	0.0000	0.0000	0.00	0.9588

Appendix Table 6.—*Results of an ANOVA with a split-plot, two-way factorial design testing the effects of FIRE, GAP, SPECIES and their interactions on the mean proportion of top-killed sapling with sprouts > 50 cm tall in the absence of herbivory*

Source	df	SS	*F*	*P*	R^2
MODEL	25	11.4051	2.89	0.0003	0.51
ERROR	70	11.0414			

Source	df	Type III SS	Type III MS	*F*	*P*
STAND	3	2.2414			
STAND x FIRE	3	0.2769	0.9230		
FIRE	1	1.4053	1.4053	15.22	0.0299
GAP	1	0.1717	0.1717	1.09	0.3004
SPECIES[1]	4	2.3222	2.3222	3.68	0.0089
FIRE x GAP	1	0.0004	0.0004	0.00	0.9599
FIRE x SPECIES	4	2.8111	2.8111	4.46	0.0029
GAP x SPECIES	4	0.4921	0.4921	0.78	0.5420
FIRE x GAP x SPECIES	1	0.3740	0.3740	0.59	0.6690

[1] The species included in this analysis were striped maple, red maple, sugar maple, American beech, and black cherry.

USDA FOREST SERVICE FARM WOODLANDS CASE STUDY—50 YEAR RESULTS FROM WEST VIRGINIA

David W. McGill and Thomas M. Schuler[1]

ABSTRACT.–Recognizing that fire, grazing, and poor cutting practices had compromised much productivity in farm woodlots in the early 1900s, USDA Forest Service personnel established a demonstration on the Fernow Experimental Forest (FEF) to show the feasibility of two harvesting systems that could be used to maintain or enhance stand productivity and provide private forest land owners with biennial revenue. Beginning in 1950, treatments were applied to create single-tree selection and 4 percent diameter limit cutting in two compartments. In this paper we document a shift in species composition, relatively equal volumes in removals, and generally increasing growing stock levels over the course of the intervening 50 years.

During the latter part of the 19th century and the first half of the 20th century, forests in the United States—particularly in the east—underwent landscape scale heavy cutting. In virtually all regions this cutting often resulted in hundred or even thousands of contiguous acres being cut without regard to the environmental consequences of these actions.

This widespread cutting, grazing, and burning, that took place during this period of forest exploitation drew the attention of conservationists throughout the United States. Fifty years ago, a group of foresters concerned with deleterious logging practices they were seeing in West Virginia, established a demonstration area in Tucker County, at what was then the recently established Fernow Experimental Forest. The intent of the demonstration area was to show farmers and other woodland owners the economic value of managing their tracts of timber, and to demonstrate silvicultural methods that could be used to provide biennial or annual revenue while assuring sustained wood production.

Today, now that the cutover stands of past centuries have regenerated and have developed into stands with valuable hardwood sawtimber, harvesting is again progressing through the forests of West Virginia and other states in the central Appalachians. Over the past decade three mills, each requiring over a million tons of hardwood pulpwood per year for composite wood products, have been established in central and southern West Virginia adding to the demand for wood products. A "mid-cycle" inventory, sponsored by the West Virginia Division of Forestry (WVDOF 1997) found that between 1989 and 1995 the growth to drain ratio—a metric that describes the proportion of wood volume growing in the forest to that cut from the forest—decreased from 3.41 in 1989 to 1.34 in 1995. Growth to drain ratios below 1 indicate harvests exceed growth and would not be sustainable for long.

These indications of accelerated timber harvesting are of particular concern in light of the 1995 timber harvest inventory reported by Fajvan and others (1998). In their assessment of 101 harvest areas, the researchers document the seeming lack of silvicultural prescriptions and intent for future stand conditions. Despite the resiliency of the forest that was seen over the past century, changing social and environmental conditions may have more negative impacts to the productivity of the forest today than in the past.

Given our current social and environmental setting, conveyance to forest land owners of information that expresses the benefits of forest management is crucial for promoting renewable natural resources production from

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the forests of today. This paper documents the 50-year productivity of two demonstration areas in order to improve understanding of long-term uneven-aged silviculture in the central Appalachian region. While the individual practices in each demonstration unit were not replicated, the case study as presented here can serve as an example of long-term management for new and experienced forest land owners, and can also stimulate hypotheses regarding possible stand trajectories and volume production under these and alternative scenarios.

STUDY AREA

Data for this case study have been collected since 1950 from Compartments 13 and 21 on the Fernow Experimental Forest (39.03° N, 79.67° W) in north-central West Virginia. The Fernow is located in the Allegheny Mountains section of the Central Appalachian Broadleaf Forest (McNab and Avers 1994). The vegetation is classified as mixed-mesophytic (Braun 1950) and dominant overstory species often include northern red oak (*Quercus rubra* L.), yellow-poplar (*Liriodendron tulipifera* L.), and sugar maple (*Acer saccharum* Marsh.) on mesic sites and red maple (*A. rubrum* L.) and chestnut oak (*Q. prinus* L.) on more xeric sites. Compartments 13 and 21 have characteristics of both mesic and xeric conditions, but overall are classified as good growing sites and have red oak site indices 50 of about 70.

In the original study plan, Holcomb and Weitzman (1950) described Compartment 13 (31 acres) as an unmanaged forest which had been high-graded and burned but retained some of the old residuals not removed in the original logging (c. 1905–1912). Compartment 21 represented even poorer starting conditions than 13 according to the initial investigators. They described Compartment 21 (31 acres) as essentially clearcut during the original logging and severely burned following logging. Consequently, in 1950 it was in an earlier stage of development with noticeably less stocking than that found in Compartment 13. These two compartments were selected for this case study because they were fairly representative of the range of farm woodland conditions found throughout the area in the early 1950s, and both were readily accessible for demonstration.

SILVICULTURAL TREATMENTS

The original intent of silvicultural treatments for both compartments was to determine the economic and silvicultural feasibility of annual harvest cuts from small areas. The initial cuts were designed to remove primarily cull trees and the most defective merchantable trees. Succeeding cuts were designed not to exceed growth until the desired structure and stocking were achieved, then removals would approximate growth. Goal diameter distributions were to approximate uneven-age and uneven-size class characteristics, although explicit silvicultural guidelines were deemed secondary to mimicking practices thought feasible by small landowners. For example, one year the harvest might remove material deemed suitable for mine timbers by removing low quality stems of less desirable species. Preferred species for retention included sugar maple, northern red oak, white oak (*Q. alba* L.), yellow-poplar, white ash (*Fraxi-nus americana* L.), and black cherry (*Prunus serotina* Ehrh.). Less desirable species were red maple, chestnut oak, birch (*Betula* sp.), beech (*Fagus grandifolia* Ehrh.), hickory (*Carya* sp.), and cucumber-tree (*Magnolia acuminata* L.). Undesirable species included blackgum (*Nyssa sylvatica* Marsh.), sassafras (*Sassafras albidum* (Nutt.) Nees), sourwood (*Oxydendrum arboreum* (L.) DC.), and black locust (*Robinia pseudoacacia* L.).

During the 1960s, each compartment was further divided into five sub-compartments of approximate equal size. Compartments were cruised on 10-year intervals where all stems greater than 5 inches in diameter at breast height (dbh) were recorded by species and sampled for hardwood tree grade. Cutting guidelines for the next 10 years would then be based on this inventory. The cutting cycle for each subcompartment was 10 years but the dates were scheduled so that one subcompartment from either Compartment 13 or 21 was marked and harvested annually. Annual harvests alternate between the two main compartments. The small size of the annual harvests was desirable, in part, because the scale would approximate the level of work the initial investigators thought some farmers could do working independently. The annual cut would also allow woodland owners to respond to changing market conditions and provide an annual cash flow.

After the initial salvage logging objectives were met, silvicultural guidelines became more explicit. Since 1974, Compartment 13 has been managed using a single-tree selection system. Initially, residual stand structure goals following harvesting were defined by inherent differences in site productivity. By 1981, a single standard was applied using a Q of 1.3 (which refers to the number of trees in successively smaller two inch diameter classes) (Nyland

1996), a residual basal area of 85 ft^2 ac^{-1} for trees 5 inches dbh and larger, and a maximum dbh of 32 inches.

Beginning in 1998, each sub-compartment was cruised separately so that more realistic marking guidelines could be determined for each sub-compartment. Consequently, we found some sub-compartments to be substantially above stand structure goals and others to be in deficit. One sub-compartment (13B1) was not harvested as scheduled after the 2000 growing season because most diameter classes were slightly below residual stand structure goals. This was a result of previously using compartment-wide characteristics to determine surplus trees in each sub-compartment. Growth projections indicate the 13B1 will have adequate volume above the residual stand structure goal at the next cutting cycle in the year 2010.

Since the early 1970s, Compartment 21 has been managed using a procedure for selection marking based on variable rates of return that are influenced by tree quality, species, and size (Trimble and others 1974). Compartment 21 is managed using the guidelines for site index 70 using a 4 percent rate of return. In essence, yellow-poplar, beech, white oak, and chestnut oak are marked when they reach the 18 inch dbh class, black cherry and white ash are marked when they reach the 20 inch dbh class, and red oak and red and sugar maple are marked when they reach the 22 inch dbh class. Additionally, this system prioritizes removal of smaller trees that are of low quality or vigor, and all short-lived species. Hereafter, we refer to this procedure as the 4 percent diameter limit. In practice, the forester can adjust rates of return with each cutting cycle to control residual stocking, although that was not done in this case (Miller and Smith 1993).

Volume estimates used in our analysis were generated from local volume tables developed from sites throughout the Fernow Experimental Forest. Originally volume tables were constructed for each compartment. In 1971, board foot volume tables were revised based on site index and species. All board foot volumes presented in this analysis were calculated based on the 1971 tables.

RESULTS

Fifty years of measurement and observation in the two experimental management compartments have provided an image of long-term potential of forest stands in the Central Hardwood region. From the standpoint of providing sustained wood production and periodic income for farm woodland owners, the two systems have similarities and some differences.

Single-Tree Selection System

Initial Composition

At the outset of this study, Compartment 13 contained an estimated total volume of 10,444 bf ac^{-1}. Two-thirds of the stems measured, however, were poletimber-sized trees ($5 \leq$ dbh ≤ 11 inches); consequently, stand structure was well characterized by a negative exponential diameter distribution. Absolute stem density (dbh ≥ 5 inches) was 153 trees per acre, comprised predominately of yellow-poplar (19 percent), beech (12 percent), birch (12 percent), and red oak (12 percent). Volume in the stand was dominated by yellow-poplar (18 percent), beech (16 percent), sugar maple (15 percent), and red oak (11 percent).

Volume Production

By the end of the first decade, standing merchantable volume had increased 811 bf ac^{-1} (9 percent) from an initial 9,240 bf ac^{-1}, and attained 10,051 bf ac^{-1} despite the relatively heavy cutting over the decade that was necessary to remove unproductive culls. During the second and third decades removals diminished, but standing volume continued to increase to 12,045 bf ac^{-1} (20 percent) and 14,209 bf ac^{-1} (18 percent), respectively. The remeasurement data from the end of the fourth decade shows that standing volume decreased 176 bf ac^{-1} (-1 percent), but again increased 991 bf ac^{-1} (7 percent) to 15,024 bf ac^{-1} by the end of the fifth decade.

Cull and Merchantable Removals

Management of this compartment under the silvicultural regime described above resulted in five decades of wood production that generated about 2,600 bf ac^{-1} per decade in merchantable removals. In reality, merchantable removals varied by decade, with less volume removed in the second and third decades than greater volumes removed in the subsequent two decades as stand structure goals were met and surpluses were removed (fig. 1).

In this compartment, initially 1,204 bf ac^{-1} of cull wood were present in the stand and through the first decade 1,249 bf of cull material were removed (table 1). At the 10-year remeasurement, only 169 bf ac^{-1} of cull and 77 bf ac^{-1} of dead material were recorded, illustrating the intent to improve growing-stock quality. The

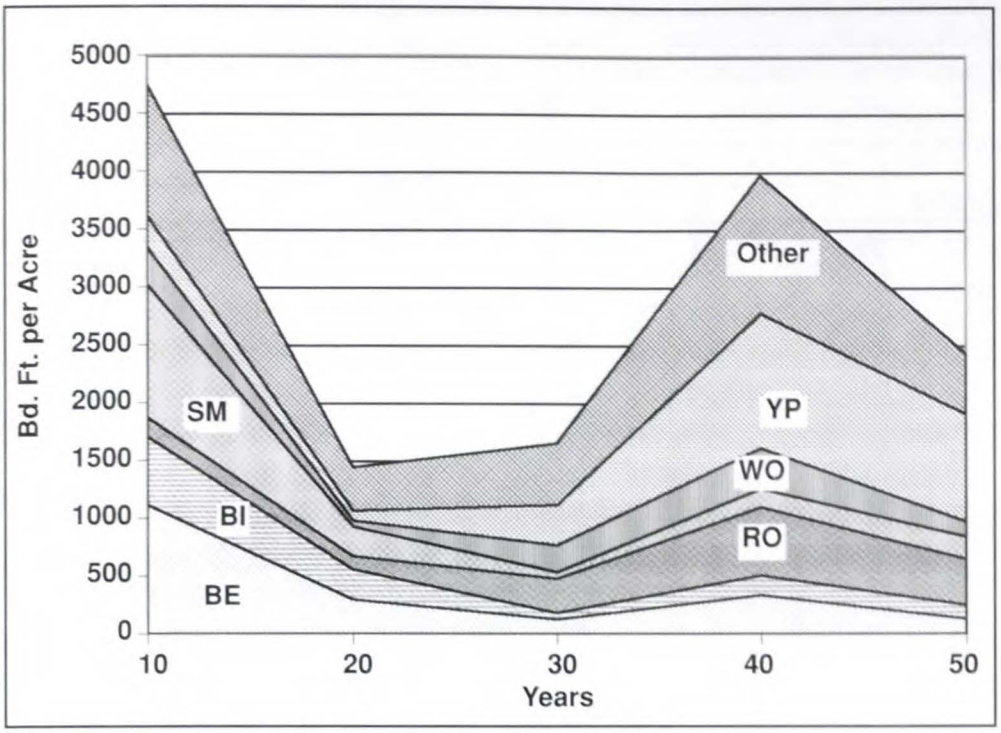

Figure 1.—*Total board foot per acre removals from single-tree selection treatment by species group and years following establishment of the Farm Woodlands Study areas at the Fernow Experimental Forest. "Years" represents the decade that the removals were made, e.g., 10 is the first 10 years of the study, 20 is the second 10 years, and so on. Legend codes are: YP=yellow-poplar, WO = white oaks, SM = sugar maple, RO = red oaks, BI = birch, BE = American beech, Other = composite of species described in methods.*

unmerchantable material that was cut throughout the first decade represented 26 percent of the removals and 12 percent of the initial standing volume in this period. The relatively high proportion of culls removed permitted the more valuable stems to acquire main canopy growing space and to accelerate their growth. This process of cull removal consequently increased potential merchantable volume production at the stand level; by the end of the second decade there were few cull trees occupying productive growing space.

Beech, birch, and sugar maple comprised the bulk of the volume removed in the first decade from the single-tree selection area (fig. 1). Of this total volume, beech was the dominant cull species (48 percent of cull volume) and sugar maple dominated the merchantable class (30 percent of merchantable volume). Sugar maple was not a preferred species during the initial logging (c. 1905–1912), which explains its importance in merchantable classes during the 1950s.

Over the next several decades this cutting pattern changed (fig. 1). With time, removals became increasingly comprised of the oaks and yellow-poplar, reflecting the intent to manage for these species by retaining vigorous individuals and giving them additional growing space. By the fourth and fifth decades, at least 30 percent of merchantable removals were yellow-poplar stems, reflecting the shifting canopy dominance patterns in the cutting units. Red oak removals, too, had increased by the fourth decade following establishment and comprised about 15 percent of the removals during the last two decades. By the end of this 50-year period, cull material in this stand was virtually nonexistent.

Shift in Stand-Level Species Composition

Species composition and stand structure in the single-tree selection area has changed throughout the course of the first 50 years of observation. Initially, yellow-poplar, beech, sugar maple, and red oak were the only species accounting for more than 10 percent of the total standing volume with yellow-poplar exceeding all others

Table 1.—*Fifty-year volume production of two Farm Woodlands Study demonstration stands on the Fernow Experimental Forest in eastern West Virginia. Volume units denote average per acre board foot volume (International 1/4-inch rule) for all stems greater than 11 inches diameter at breast height. The study was initiated in 1949.*

Volume component	Years since establishment					
	0	10	20	30	40	50
	Single-tree selection system					
Growing stock	(bf/ac)	(bf/ac)	(bf/ac)	(bf/ac)	(bf/ac)	(bf/ac)
Merchantable	9,240	10,051	12,045	14,209	14,033	15,024
Cull	1,204	169	44	48	0	43
Dead	0	77	19	36	77	107
Removals						
Merchantable		3,491	1,335	1,611	3,961	2,427
Cull		1,249	99	36	5	0
Cumulative yield [1]	9,240	13,542	16,871	20,646	24,431	27,849
	4% diameter limit cutting					
Growing stock						
Merchantable	5,366	6,108	9,097	10,443	11,251	12,204
Cull	697	105	75	47	54	19
Dead	0	33	43	42	151	102
Removals						
Merchantable		2,204	496	3,377	3,931	2,829
Cull		822	40	33	76	22
Cumulative yield	5,366	8,312	11,797	16,520	21,259	25,041

[1]Cumulative yield for a given period includes initial merchantable volume, the change in merchantable volume, and the volume of merchantable removals.

(18 percent). After 50 years, yellow-poplar accounted for 36 percent of the volume followed by red oak with 26 percent. No other species accounted for more than 8 percent.

The change in composition has not stopped. Despite the fact that yellow-poplar and red oak have grown to dominate stand composition, the increasingly larger proportion of these species in the harvest volume, combined with their lack of ingrowth into smaller size classes indicates a future depletion of these species from this unit (fig. 2). Moreover, shade tolerant species like American beech, red maple, and sugar maple have increasingly dominated ingrowth into the smaller size classes (fig. 3) and will likely begin to dominate merchantable harvest volumes in the next half-century.

4 Percent Diameter Limit Cutting

Initial Composition

In Compartment 21, initial volume was lower than in Compartment 13 because past logging and logging-era fire left fewer residual trees. Still, just over two-thirds of the stems measured were poletimber-sized and stand structure was similarly well characterized by a negative exponential diameter distribution. Absolute initial stem density was 200 stems ac^{-1} and comprised mostly of red oak (18 percent), birch (16 percent), yellow-poplar (13 percent), and beech (10 percent). Initial board foot volume was largely beech (28 percent), yellow-poplar (13 percent), red oak (15 percent), and sugar maple (10 percent).

Volume Production

Initial standing merchantable volume (5,366 bf ac^{-1}) in this compartment was only 58 percent of that in Compartment 13 (table 1). Yet, periodic harvesting of the two compartments during half a century generated almost identical volumes.

Using the 4 percent diameter limit selection system, this compartment exhibited positive increases in standing merchantable volume in each of the five decades of this study. By the end of the first decade, merchantable volume had increased 742 bf ac^{-1} (14 percent) to 6,108 bf ac^{-1} from its initial volume. Standing volume

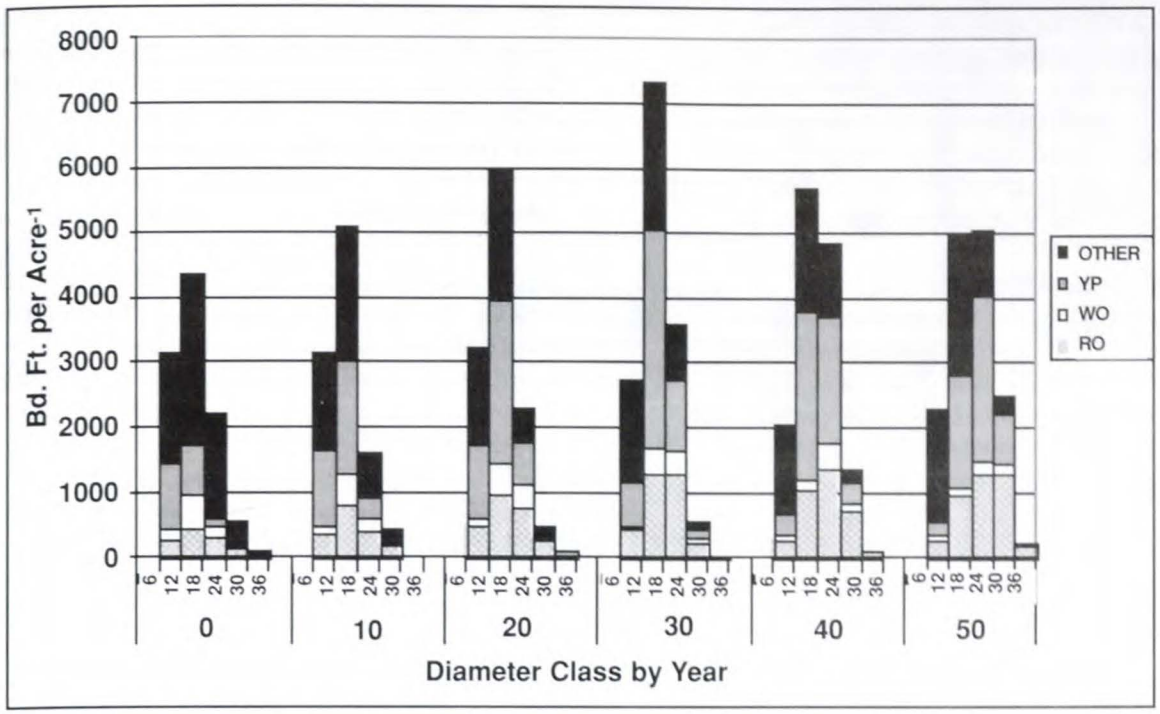

Figure 2.—*Volume distribution (International 1/4-inch rule) in single-tree selection area, highlighting red oak, white oak, and yellow-poplar by years following establishment of the Farm Woodlots Demonstration at the Fernow Experimental Forest near Parsons, West Virginia. Legend codes are: YP = yellow-poplar, WO = white oaks, RO = red oaks, and Other = composite of species as described in Methods.*

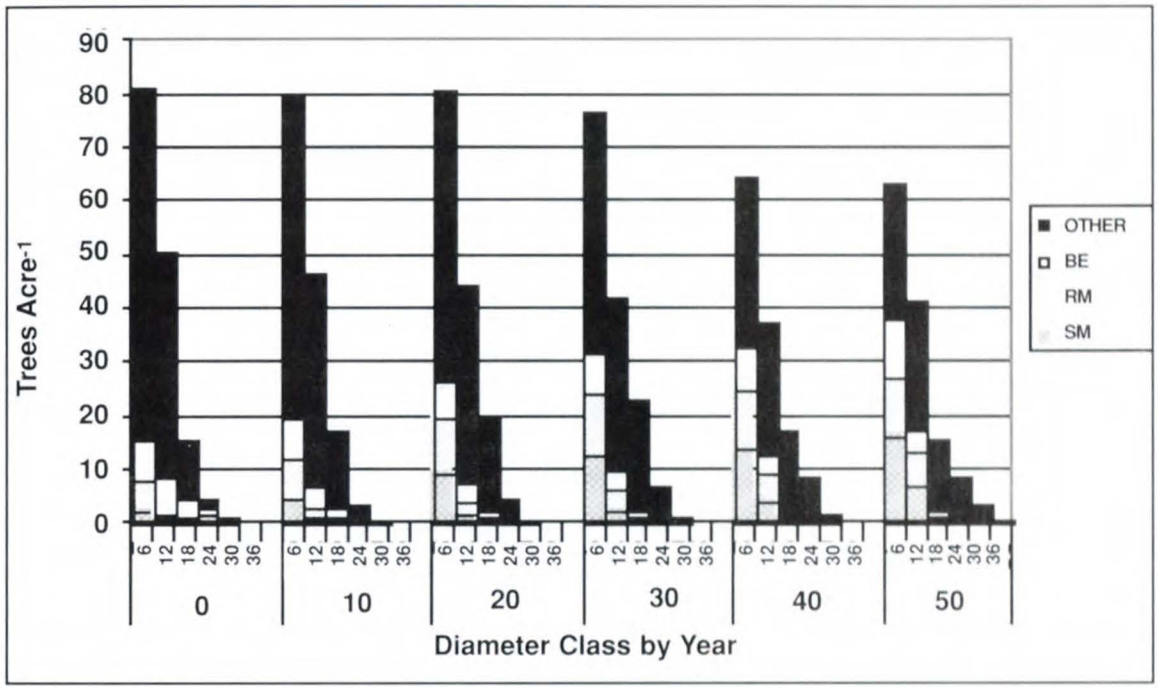

Figure 3.—*Frequency distribution (trees per acre) in single-tree selection area, highlighting shade tolerant species, by years following establishment of the Farm Woodlots Demonstration at the Fernow Experimental Forest near Parsons, West Virginia. Legend codes are: SM = sugar maple, RM = red maple, BE = American beech, and Other = composite of species as described in Methods.*

increased to 9,097 bf ac^{-1}—a change of 49 percent—during the second decade when a pulse of poletimber grew into small sawtimber-sized trees. Over the next three decades standing merchantable volume increased 15 percent, 8 percent, and 8 percent, respectively, resulting in 12,204 bf ac^{-1} at the end of the fifth decade.

Cull and Merchantable Removals

Of the initial 6,063 bf ac^{-1} measured in this compartment, 697 bf ac^{-1} (11 percent) were cull. During the first decade, 822 bf ac^{-1} of cull material were removed. Fifty-five percent of this volume was beech between 12 and 24 inches dbh and 22 percnt was red maple mostly greater than 16 inches dbh. This compartment was rapidly cleared of its unmerchantable and undesirable growing stock. Table 1 attests to this fact and shows very little cull volume following the first decade.

Species composition of removals changed through time in this stand. Initially, the majority of the volume removed was beech, but composition of removals became mostly red oak and yellow-poplar in the later cutting cycles (fig. 4).

Proportions of these two species relative to the total volume removed in five decades from the beginning of the study to the end of year 50 were: 7 percent, 13 percent, 59 percent, 69 percent, and 66 percent.

Shift in stand-level species composition

Continuing change in species composition in this compartment is evident, given the diminution of shade intolerant species ingrowth into smaller diameter classes (fig. 5). Further evidence of this compositional shift is found in the increasing number and dominance of shade tolerant species in these smaller size classes (fig. 6).

DISCUSSION

At a minimum, these two compartments of the Farm Woodlands study have provided a long-term view of the behavior of forest communities as forest management guides them into selection systems.

In both systems, the common and consistent practice of first removing all undesirable (or unacceptable) growing stock and applying the protocols of each system, led to a period during

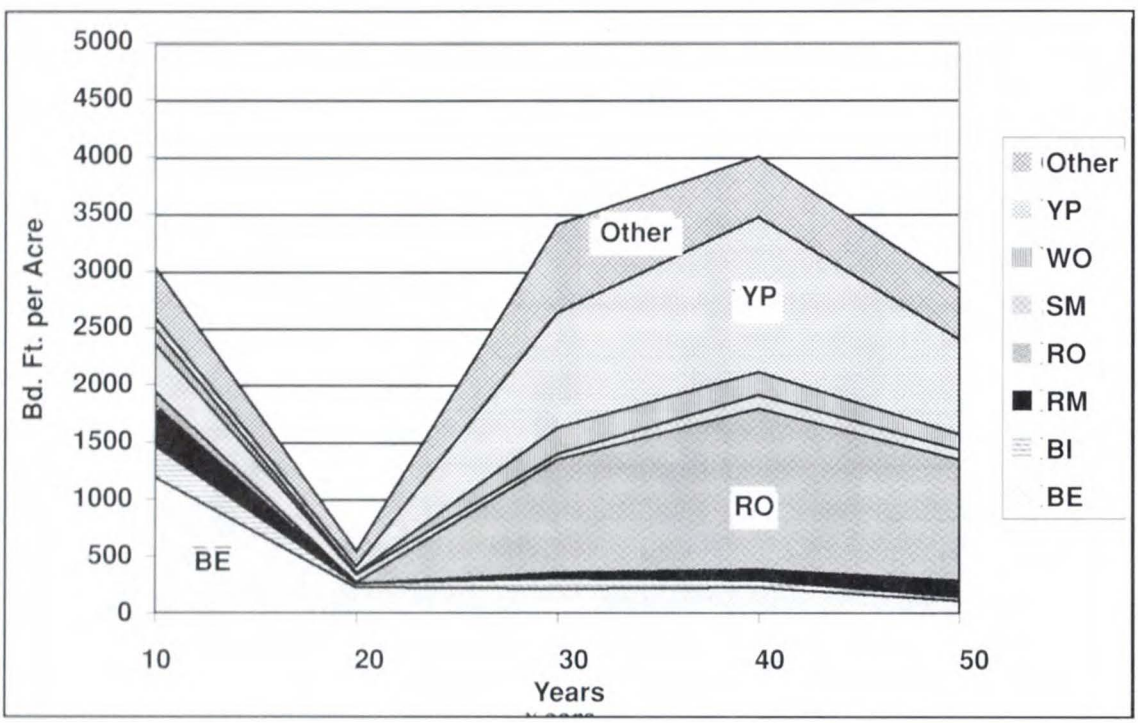

Figure 4.—*Total board foot per acre removals from 4 percent diameter limit cutting by species group and years following establishment of the Farm Woodlands Study areas at the Fernow Experimental Forest. "Years" represents the decade that the removals were made, e.g., 10 is the first 10 years of the study, 20 is the second 10 years, and so on. Legend codes are: YP=yellow-poplar, WO = white oaks, SM = sugar maple, RO = red oaks, RM = red maple, BI = birch, BE = American beech, and Other = composite of species as described in Methods.*

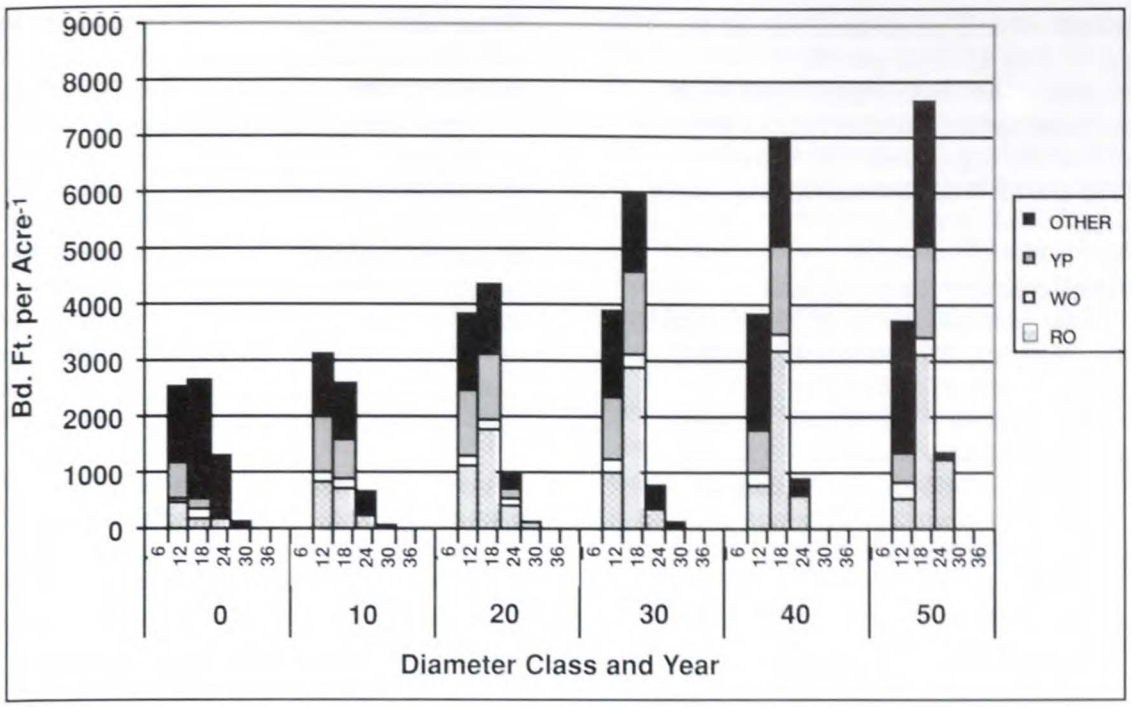

Figure 5.—*Volume distribution (International 1/4-inch rule) in 4 percent diameter limit cutting, highlighting red oak, white oak, and yellow-poplar by years following establishment of the Farm Woodlots Demonstration at the Fernow Experimental Forest near Parsons, West Virginia. Legend codes are: YP = yellow-poplar, WO = white oaks, RO = red oaks, and Other = composite of species as described in Methods.*

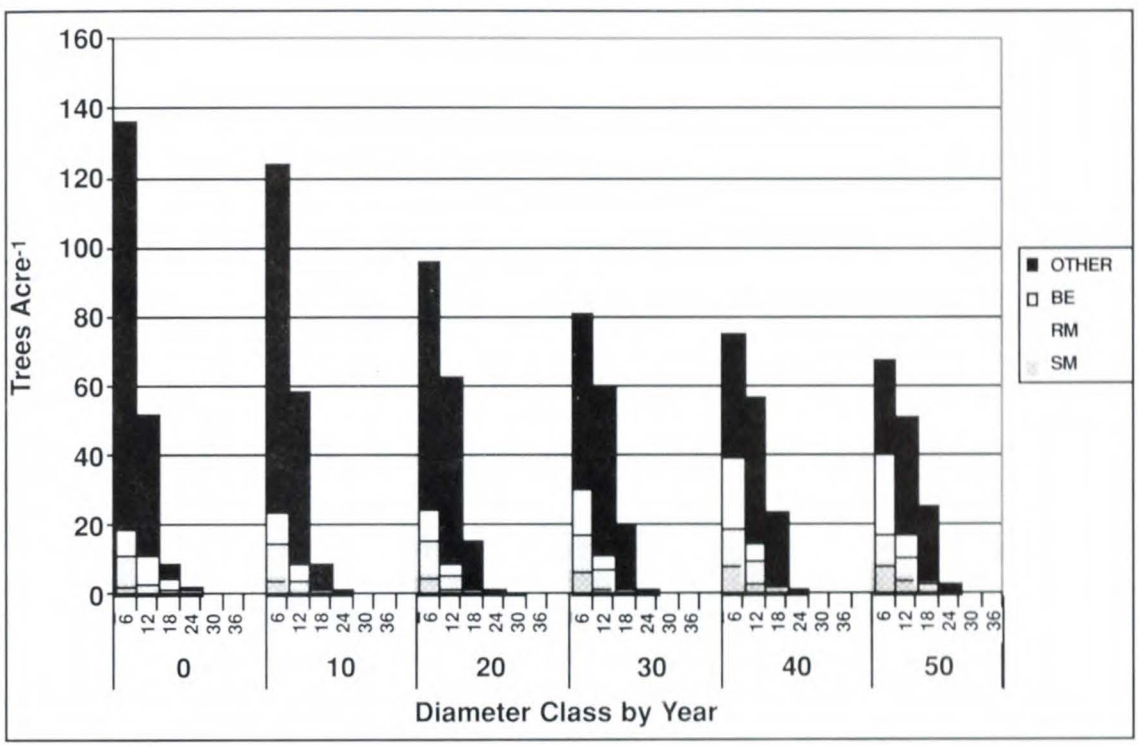

Figure 6.—*Frequency distribution (trees per acre) in 4 percent diameter limit cutting, highlighting shade tolerant species, by years following establishment of the Farm Woodlots Demonstration at the Fernow Experimental Forest near Parsons, West Virginia. Legend codes are: SM = sugar maple, RM = red maple, BE = American beech, and Other = composite of species as described in Methods.*

the second decade where removals were at a minimum (figs. 1 and 4). This period of decreased removals was less pronounced in the single-tree selection system where higher initial volumes provided the opportunity to remove merchantable material without sacrificing stand structure goals.

Both silvicultural systems have produced remarkably similar volumes through the second half of the 20th century. As mentioned earlier, total removals were virtually identical and averaged only 2 bf ac^{-1} $decade^{-1}$ difference between the two units (2,567 bf ac^{-1} $decade^{-1}$ removed in Compartment 13 and 2,565 bf ac^{-1} $decade^{-1}$ removed from Compartment 21). Average gross and net productivity also have similarly close values (table 2). Interestingly, while the 4 percent diameter limit cutting started with just over 3,800 bf ac^{-1} less standing merchantable volume than that initially in the single-tree selection system, by the end of 50 years—and with nearly equal removals over the period—the gap between standing merchantable volumes had narrowed to a difference of about 2,800 bf ac^{-1}. Indeed, over this period the net periodic annual increment (PAI) in the 4 percent diameter limit averaged 394 bf ac^{-1} yr^{-1}, nearly a 6 percent higher PAI than that of the single-tree selection system.

When considering the similarities in productivity and species composition that both systems have produced, the greater complexities of stand management using single-tree selection guidelines hardly seem justified. Miller and Smith (1993) provide additional suggestions for using the flexible diameter-limit approach and recommend it as a practical alternative to single-tree selection based on other long-term studies. Moreover, small private woodland owners are unlikely to have the skills to apply a single-tree selection system, but may be able to employ many of the ideas of the flexible diameter limit system.

What might these systems have produced if left unmanaged? A rough comparison can be made to a nearby compartment that serves as an unmanaged reference stand for the Fernow Experimental Forest. Watershed 4B (WS4B) (13 acres) has been measured five times in the last 48 years using four, 100-percent cruises and one fixed radius plot sample. These data provide a mirror against which we can cautiously compare managed stands. Table 3 shows the yields achieved on the 13-acre reference area beginning in 1959 at approximately the same time as the year 10 inventories were made on the Farm Woodlands study. At the initial measurement in 1959, WS4 contained 9,325 bf ac^{-1} of merchantable sawtimber. At the comparable

Table 2.—*Growth and yield values from two selection systems of the Farm Woodlot Study Compartments located on the Fernow Experimental Forest near Parsons, West Virginia. Both gross and net increments express compartment growth including ingrowth. Gross increment treats cull volumes cut during harvesting operations as mortality.*

	Years						
Growth Parameter	**0**	**10**	**20**	**30**	**40**	**50**	**Mean**
			Single-tree selection system				
Yield	9,240	13,542	16,871	20,646	24,431	27,849	
Gross Increment		5,551	3,428	3,811	3,790	3,418	
Gross PAI		555	343	381	379	342	*400*
Δ GPAI			-38%	11%	-1%	-10%	
Net Increment		4,302	3,329	3,775	3,785	3,418	
Net PAI		430	333	378	379	342	*372*
Δ NPAI			-23%	13%	0%	-10%	
			4 percent diameter limit cutting				
Yield	5,366	8,312	11,797	16,520	21,259	25,041	
Gross Increment		3,768	3,525	4,756	4,815	3,804	
Gross PAI		377	353	476	482	380	*413*
Δ GPAI			-6%	35%	1%	-21%	
Net Increment		2,946	3,485	4,723	4,739	3,782	
Net PAI		295	349	472	474	378	*394*
Δ NPAI			18%	36%	0%	-20%	

measurement, the single-tree selection system had 10,051 bf ac^{-1} and the 4 percent diameter limit had 6,108 bf ac^{-1}, or 726 bf ac^{-1} more and 3,217 bf ac^{-1} less than WS4B, respectively.

At its final measurement, which occurred in 2001, WS4 contained 20,677 bf ac^{-1} in merchantable volume and 2,789 bf ac^{-1} in dead sawtimber volume. Even if the assumption that the dead material was still recoverable, its yield at that time would be 23,466 bf ac^{-1}. In both of the managed systems in this study, yield exceeded that of WS4B by 4,383 bf ac^{-1} in the single-tree selection system and by 1,575 bf ac^{-1} in the 4 percent diameter limit.

CONCLUSIONS

Given the initial structure, composition, and silvicultural regimes applied to these compartments, species composition shifts have occurred over the study period. Yellow-poplar and red oak, once existing among larger maples and beech, now dominate the site in the larger size classes. The compositional shift will continue as the larger yellow-poplar and oaks are harvested and replaced by more shade tolerant maples and beech that are now dominating the pole-timber-size classes. This shift will likely occur more quickly in the 4 percent diameter limit system because the oaks and yellow-poplar can be cultured throughout the size distribution in the single-tree selection system to a maximum of 32 inches dbh.

The silvicultural activities that aimed to bring these compartments under management aggressively eliminated cull trees from the stands. This consequently served to maximize merchantable volume production on the crop trees, evidenced by very little or no cull volumes in the later remeasurements and harvest data. Under these management scenarios, stand transition from states of unspecified management to uneven-aged silvicultural systems, the volume of removals in both compartments was relatively high in the first decade and had the lowest levels in the second decade. This is related to ridding the stand of undesirable trees during the first decade of management and then allowing the stands to accumulate growing stock during the second decade of management. This general pattern can be expected when applying these partial cutting practices to previously unmanaged and exploited forest stands.

The flexible diameter system is a practical alternative to the complexities of single-tree selection. When applied appropriately, similar outcomes in productivity and species composition can be expected.

Both systems evaluated here lead to dominance of shade tolerant species. In our case, sugar maple provides a valuable shade tolerant commercial species. However, relative values of red maple and beech are much less and landowners should be aware of the potential of ultimately changing periodic revenues even while periodic productivity may not be drastically altered. Moreover, American beech is suffering the effects of beech bark disease and is likely to be limited to predominantly understory status in many areas.

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During the past 50 years many individuals have played important roles in this case study. We especially want to acknowledge the pioneering

Table 3.—*Sawtimber volumes from unmanaged (control) Watershed 4 at the Fernow Experimental Forest near Parsons, West Virginia. Dates of remeasurement roughly correspond to 10, 20, 30, 40, and 50 year remeasurements of the Farm Woodlands Study established in nearby compartments. Volumes are expressed as board feet (International 1/4-inch rule) per acre. Yield assumes dead material is merchantable.*

	Year of measurement				
	1959	1972	1984	1994‡	2001
Years in interval		14	11	10	7
Volumes					
Merchantable	9,325	13,149	16,942	21,351	20,677
Dead	-	369	1,202	2,629	2,789
Cull	490	698	325	-	-
Yield	9,325	13,518	18,144	23,980	23,466
Percent Dead and cull	5%	8%	8%	11%	12%

efforts of C.J. Holcomb and Sidney Weitzman in establishing this study; G.R. Trimble, Jr. for his continuation of their initial concepts; H.C. Smith for establishing the more explicit silvicultural guidelines used to achieve the initial management objectives; G.M. Miller for carefully documenting and applying prescriptions; R. Rosier and R. Hovatter for much of the cruising and marking over the past several decades; and F. Wood and L. Plaugher for database management. We also want to thank the Fernow logging crew, which has done all of the logging for this study for the past half-century. Current member of this crew include R. Poling, M. Owens, and S. Hockman.

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A COMPARATIVE ANALYSIS OF THE DIVERSITY OF WOODY VEGETATION IN OLD-GROWTH AND SECONDARY SOUTHERN APPALACHIAN COVE FORESTS

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ABSTRACT.—Characteristics of woody vegetation were compared across six different southern Appalachian cove forests. Trees greater than 6.35 cm dbh were point sampled and regeneration was tallied on 25 m^2 subplots at all study sites. Overstory composition and structure differed between secondary and old-growth sites, which were dominated by yellow-poplar and eastern hemlock, respectively. Mean species richness and density-based diversity indices were higher in the secondary sites for both the overstory and regeneration strata. Shade tolerant species were favored in the regeneration strata at all sites, and yellow-poplar regeneration was rare throughout the study.

Southern Appalachian mixed mesophytic or cove forests occupy valleys and lower slopes at low elevations, spreading outward on sheltered and northerly slopes at higher elevations to around 1,400 meters (Whittaker 1956). Old-growth remnants of this forest type are commonly recognized as some of the most diverse of any in the eastern United States (Martin 1992), in which many characteristic tree species have been noted to grow to record sizes (Stupka 1960). The vegetational diversity of the more commonly found second-growth cove forests in this region is not as widely documented.

The intermediate disturbance hypothesis (Connell 1978) states that the highest levels of diversity in forest communities often occur in intermediate seral stages after a large-scale disturbance event where the presence of both pioneer and climax-associated species overlaps. If many of the second-growth cove forests in the southern Appalachians are indeed in an intermediate stage of succession (Busing 1998), the diversity of these second-growth stands may, in fact, be higher than their old-growth counterparts. It was therefore the objective of the present study to clarify this issue by surveying the woody vegetation (both overstory and regeneration) in several representative stands each of old-growth and second-growth southern Appalachian cove forests and comparatively analyzing them in terms of diversity and other stand-level attributes.

Great Smoky Mountains National Park contains perhaps one of the largest contiguous tracts of old-growth forest in the eastern United States (NPS 1981, Kemp 1993, White and others 1996), and estimates of actual old-growth within the park vary from 16 to 40 percent (Pyle 1985). However, a large portion of these estimates includes high elevation spruce-fir forests, and old-growth mixed mesophytic communities are fairly limited in extent even in protected areas within this region such as the National Park (Martin 1992). According to Whittaker (1956) anthropogenic disturbances in this mountainous region (e.g., logging, farming, and other settlement-related activities) were heavily concentrated in lower elevations due primarily to accessibility, and undisturbed areas in lower valleys are relatively rare. The establishment of the national park in the 1930s resulted in the cessation of such activities, and heavily disturbed areas within park boundaries were subsequently left to recover through natural processes. Protected natural areas such as this have since afforded excellent opportunities to study and monitor the process of natural succession in mixed mesophytic forests of this region. The existence of old-growth remnants of the same forest type in such close proximity provides an integral source of baseline data against which attributes of younger, second-growth stands can be evaluated (Martin 1992).

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Southern Appalachian mixed mesophytic communities can contain about 25-30 tree species with up to eight occurring as dominants or codominants, including eastern hemlock (*Tsuga canadensis* (L.) Carr.), yellow buckeye (*Aesculus octandra* Marsh.), white basswood (*Tilia heterophylla* Vent.), Carolina silverbell (*Halesia tetraptera* Ellis), sugar maple (*Acer saccharum* Marsh.), yellow birch (*Betula alleghaniensis* Britton), yellow-poplar (*Liriodendron tulipifera* L.), and American beech (*Fagus grandifolia* Ehrh.). Composition as well as dominance may vary between different stands, and not all of these eight species necessarily occur at any one given site (Braun 1950). More detailed descriptions of the forest type can be found in Braun (1950), Whittaker (1956), and Martin (1992).

Natural disturbances in cove forests consist primarily of relatively small-scale single or multiple-tree canopy gaps created by windthrow or senescence of large individuals comprising the forest canopy (Lorimer 1980, Runkle 1982, Busing 1994). The importance of canopy gap dynamics in old-growth cove forests is well documented in the literature (Runkle 1981, Runkle and Yetter 1987, Busing and White 1997). These and other studies (Buckner and McCracken 1978, Barden 1989, Busing 1989, Runkle 1998) indicate that these forests are more or less at equilibrium compositionally, and that canopy gap dynamics are, and have been, sufficient in the perpetuation of this compositional stability over time.

Second-growth cove forests in the southern Appalachians are often heavily dominated by yellow-poplar, which is a pioneer species and rapidly colonizes abandoned agricultural or cut over lands (Buckner and McCracken 1978). Yellow-poplar is relatively long lived and may persist as the dominant canopy species in successional forests for up to about 150 years, and an additional 150-250 years may be required for the species to decline in abundance to levels found in old-growth forests in this region (Busing 1995).

Although the diversity of second-growth cove forests has not been documented as extensively in the literature as old-growth cove forests, the issue has been addressed. Clebsch and Busing (1989) found that mid-successional cove forests had greater stem densities and tree species richness than old-growth forests, and linked this to the co-occurrence of shade intolerant species in the overstory and shade tolerant species in the understory. Clebsch and Busing (1989) also reported that while Shannon-Wiener (H') diversity indices were higher for old-growth forests than second-growth forests when those indices were based on biomass, no clear trends were apparent when they were based on densities, although the highest value reported was for a second-growth forest.

Busing (1998) suggests that second-growth cove forests in the southern Appalachians may persist in the transition phase anywhere from 100-300 years before reaching the final steady state phase in Bormann and Likens (1979) four-phase model of forest succession, and furthermore that little is known about the dynamics of the transition phase in cove forests. The results of the present study therefore address not only the issue of diversity as it relates to successional and old-growth cove forests, but also provide a timely source of benchmark data pertaining to second-growth cove forests as they approach the long transition phase of their development. Improved understanding of the natural dynamics of successional yellow-poplar stands may also prove beneficial in the management of this economically important timber species.

METHODS

Three old-growth and three second-growth southern Appalachian cove forests were chosen for the study and sampled in the fall of 1996 and the summer of 1997 (table 1). All of the study sites are located within Great Smoky Mountains National Park (GSMNP) except Little Santeetlah Creek, which is located in Joyce Kilmer Memorial Forest (JKMF), North Carolina. The three old-growth sites are well noted in the literature for their old-growth attributes. More detailed descriptions of the Little Santeetlah Creek watershed and Joyce Kilmer Memorial Forest can be found in Oosting and Bourdeau (1955), Lorimer (1980), USFS (1988), and Hedman and Van Lear (1995). Detailed descriptions of the Albright Grove site can be found in Busing (1993) and Houk (1993). Descriptions of the Ramsay Cascades site can be found in Houk (1993) and Kemp (1993).

Two of the second-growth sites, Maddron Bald Trail and Middle Prong, are characterized by a diffuse settlement-related disturbance history (e.g., selective harvesting, grazing, and farming) and are located between areas of concentrated settlement at lower elevations on the fringe of the park and undisturbed areas at higher elevations (Pyle 1985). Remnants of stone fences and stone piles are present at both sites (Martin

1992). Although the aspect at the Middle Prong site is primarily to the southwest, the relatively flat topography is typical of a broad, lower elevation cove. The Curry Mountain site is located in an area classified as disturbed by greatly mechanized, highly intensive, and nonselective pre-park corporate logging operations (Pyle 1985).

The sampling procedure at each site consisted of first establishing line transects with randomly determined starting points. Compass headings were also randomly chosen whenever possible, but due to topographical constraints some transects simply paralleled the lower-lying streambeds often associated with cove forest vegetation. Sample plots were then located at evenly spaced intervals along the line transects. The number of sample plots at each site varied considerably due primarily to spatial constraints (table 1).

The overstories of the old-growth and second-growth sites were sampled using 20 factor and 10 factor prisms, respectively. Information collected for the overstory data consisted of species identification and stem diameters at breast height (dbh) for all tallied live trees greater than 6.35 cm dbh, as well as the heights of 2 dominant or codominant trees per plot. Recorded data were used to determine site species richness and compositional parameters such as density, basal area, and frequency. Importance values were calculated as the summation of relative density, relative basal area, and relative frequency. Average canopy height at each site was determined from recorded tree heights. Diversity at each site was quantified using Shannon-Wiener (H') diversity indices and corresponding equitability (E) indices, as well as Simpson's (D) index of diversity (Krebs 1994). All Shannon-Wiener functions were calculated using base 2 logarithms. Sorensen's similarity indices were used to quantify compositional similarity between sites (Mueller-Dombois and Ellenberg 1974).

Regeneration data were obtained from 25 m^2 subplots located in previously determined quadrants off of the line transect points. All live tree and shrub species greater than 30.5 cm in height and less than 6.35 cm in diameter were tallied in the subplots. Due to its multi-stemmed growth form, rosebay rhododendron (*Rhododendron maximum* L.) was recorded as percent coverage of the subplot when present. Regeneration importance values were calculated as the summation of relative density and relative frequency. Shannon-Wiener and Simpson's diversity indices as well as equitability indices were also computed for the regeneration component at each site. Sorensen's similarity indices were again used to compare each of the sites, as well as to measure the compositional similarity of tree species in the overstory and regeneration within sites. All regeneration diversity and similarity indices were calculated using only data for tree species.

RESULTS AND DISCUSSION

Overstory

Overstory site attributes are presented in table 2. Mean species richness was higher in the secondary sites (mean=19.0) than in the old-growth sites (mean=11.7). Site densities (stems ha^{-1}) were also higher in the secondary sites (mean=1014.7) than in the old-growth sites (mean=360.5). The unusually high stem density at Curry Mountain can be attributed to high numbers of several maple species and flowering dogwood (*Cornus florida* L.) in the smaller diameter classes. Basal areas (m^2 ha^{-1}) and average canopy heights were similar across all sites, with means slightly higher in the old-growth stands.

Table 1.–*Study site characteristics*

Site	Description	Elevation	Aspect	Plots	Location
		- m -			
Albright Grove	Old-growth	1,020	N	12	GSMNP
Ramsay Cascades	Old-growth	1,000	N	8	GSMNP
Little Santeetlah Creek	Old-growth	750	N	31	JKMF
Maddron Bald Trail	Second-growth	750	N	31	GSMNP
Middle Prong	Second-growth	550	SW	15	GSMNP
Curry Mountain	Second-growth	700	NE	15/13[1]	GSMNP

[1]At Curry Mountain there were 15 overstory plots and 13 regeneration subplots.

Table 2.–*Overstory attributes of old-growth and second-growth sites*

	OLD-GROWTH			SECOND-GROWTH		
Site	Albright Grove	Ramsay Cascades	Little Santeetlah Creek	Maddron Bald Trail	Middle Prong	Curry Mountain
Density ha^{-1}	456.1	258.2	367.3	546.0	590.4	1,907.6
Basal area (m^2 ha^{-1})	45.9	40.2	36.4	36.7	32.6	46.5
H′ (density)	1.62	2.48	2.90	2.01	3.46	3.21
H′ (basal area)	2.26	2.90	2.58	0.96	2.82	3.49
E (density)	0.49	0.77	0.72	0.54	0.76	0.73
E (basal area)	0.68	0.91	0.65	0.26	0.62	0.79
D (density)	0.54	0.77	0.81	0.66	0.88	0.85
D (basal area)	0.70	0.85	0.71	0.30	0.74	0.88
Species richness	10	9	16	13	23	21
Mean height of canopy trees (m)	42	47	42	39	41	38

Mean Shannon-Wiener diversity indices based upon stem densities were higher in the secondary sites (mean=2.9) than in the old-growth sites (mean=2.3), and corresponding equitability (E) indices had similar means and ranges in the old-growth and secondary sites (table 2). Simpson's index of diversity yielded the same results as the Shannon-Wiener index. When based upon basal areas, however, Shannon-Wiener diversity indices spanned a much greater range and had a slightly lower mean in the secondary sites (mean=2.4) than in the old-growth sites (mean=2.6). Corresponding equitability indices also had a greater range and lower mean in the secondary sites (mean=0.56) than in the old-growth sites (mean=0.75). Simpson's diversity indices again yielded the same results as the Shannon-Wiener indices when based on basal area.

Eastern hemlock was clearly the dominant species throughout the old-growth sites and had the highest relative density, relative basal area, and importance value at all three of them (table 3). Other species with relatively high importance values at the old-growth sites varied considerably by site, but included Carolina silverbell, red maple (*Acer rubrum* L.), white basswood, sweet birch (*Betula lenta* L.), yellow-poplar, yellow buckeye, and American beech. Other typical southern Appalachian cove forest overstory species such as sugar maple, yellow birch, and cucumbertree (*Magnolia acuminata* L.) (Braun 1950, Whittaker 1956) were relatively minor species over the three sites. American holly (*Ilex opaca* Ait.) and mountain holly (*I. montana* Torr. & Gray) were the only understory tree species tallied in any of the old-growth sites, and were only present at Little Santeetlah Creek.

Yellow-poplar was dominant in two of the three secondary sites, and a relatively important species in the third (table 4). Red maple, sugar maple, or both were consistently important species throughout the secondary sites. Sweetgum (*Liquidambar styraciflua* L.) was an important overstory species at the Middle Prong site, and northern red oak (*Quercus rubra* L.) and chestnut oak (*Q. montana* Willd.) were important overstory species at Curry Mountain. Oak and hickory species were much more prevalent in the secondary sites than in the old-growth sites. Understory tree species also occurred more commonly in the secondary sites, including flowering dogwood, striped maple (*Acer pensylvanicum* L.), musclewood (*Carpinus caroliniana* Walt.), sassafras (*Sassafras albidum* (Nutt.) Nees), and American holly. Black locust (*Robinia pseudoacacia* L.) was present in the overstory at all three secondary sites.

Overall diameter distributions of the old-growth and secondary sites are presented in figure 1. Although some irregularities are apparent, most of the site diameter distributions exhibit or approach the negative log-linear shape characteristic of uneven-aged stands (Daniel and others 1979). At the Maddron Bald Trail site there was a noticeable increase in stem density in the intermediate diameter classes.

Diameter distributions of the six most important species at each site are presented in figure 2. In the old-growth sites, the largest diameter classes were consistently occupied by yellow-poplar, although typically at low densities. Eastern hemlock consistently dominated the smaller diameter classes with a variable assemblage of other shade tolerant species, but was also

Table 3.–*Overstory composition of old-growth sites*

Species	ALBRIGHT GROVE				RAMSAY CASCADES				L. SANTEETLAH CREEK			
	RD[1]	RBA[2]	%F[3]	IV[4]	RD	RBA	%F	IV	RD	RBA	%F	IV
Tsuga canadensis	64.7	44.2	100.0	44.8	39.0	22.9	75.0	26.5	32.1	49.6	100.0	36.0
Halesia tetraptera	16.5	30.0	100.0	24.0	11.5	14.3	62.5	13.5	5.0	2.4	19.4	4.2
Fagus grandifolia	12.6	5.8	41.7	9.7	1.9	2.9	12.5	2.6	2.9	4.5	25.8	4.7
Liriodendron tulipifera	0.7	5.8	41.7	5.7	4.0	12.9	37.5	8.5	1.9	13.4	61.3	10.5
Acer rubrum	0.9	5.0	25.0	4.1	20.3	14.3	62.5	16.4	0.7	2.0	16.1	2.3
Acer saccharum	2.3	2.5	25.0	3.7	--	--	--	--	10.2	2.0	9.7	4.9
Betula alleghaniensis	1.1	2.5	25.0	3.3	2.8	5.7	37.5	5.8	3.2	0.4	3.2	1.5
Tilia heterophylla	0.5	1.7	16.7	2.1	13.6	14.3	75.0	15.2	4.0	8.5	45.2	8.1
Magnolia fraseri	0.5	1.7	8.3	1.4	--	--	--	--	1.7	1.2	6.5	1.5
Prunus serotina	0.2	0.8	8.3	1.1	--	--	--	--	--	--	--	--
Aesculus octandra	--	--	--	--	6.3	11.4	50.0	9.8	--	--	--	--
Betula lenta	--	--	--	--	0.7	1.4	12.5	1.7	13.1	9.8	51.6	12.1
Ilex montana	--	--	--	--	--	--	--	--	22.5	1.2	6.5	8.5
Magnolia acuminata	--	--	--	--	--	--	--	--	0.7	2.0	12.9	2.0
Fraxinus americana	--	--	--	--	--	--	--	--	0.2	1.2	9.7	1.3
Quercus rubra	--	--	--	--	--	--	--	--	0.1	0.8	6.5	0.9
Ilex opaca	--	--	--	--	--	--	--	--	1.2	0.4	3.2	0.8
Carya cordiformis	--	--	--	--	--	--	--	--	0.7	0.4	3.2	0.6

1 Relative density.
2 Relative basal area.
3 Percent frequency.
4 Importance value.

present at low densities in large diameter classes. In the second-growth sites, a pronounced spike in yellow-poplar density occurred in the 40 cm dbh class at Maddron Bald Trail and in the 60 cm dbh class at Middle Prong. At Curry Mountain, yellow-poplar, northern red oak, and chestnut oak dominated the intermediate and upper diameter classes. The smallest diameter class at this site was clearly dominated by sugar maple and red maple, which were also present at low densities in some of the larger diameter classes.

The Shannon-Wiener diversity index combines the two components of species richness and the proportional allocation of these species within a stand. A greater number of species and a proportional distribution of these species within a stand equates to a higher diversity index value. For example, Ramsay Cascades and Middle Prong both have relatively high density-based equitability indices, which indicates a proportional allocation of density among species in these stands (table 2). Middle Prong has over twice as many species as Ramsay Cascades, however, and therefore has greater diversity according to the Shannon-Wiener index. A similar argument holds for the Curry Mountain site, which also has high equitability and diversity indices. When the indices are based on basal area, however, the situation changes.

Basal area was not as evenly distributed among species at Middle Prong as density, and the Shannon-Wiener index value was accordingly lower. Ramsay Cascades had an even higher equitability index based on basal area than based on density, and the diversity index was accordingly greater. The relatively low diversity indices at Albright Grove and Maddron Bald Trail reflect the dominance of eastern hemlock and yellow-poplar at those sites, respectively. Basal area was more equitably distributed among species than density at Albright Grove, but the opposite was true at Maddron Bald Trail, where yellow-poplar accounted for over 80 percent of the relative basal area (table 4).

Clebsch and Busing (1989) also reported higher Shannon-Wiener diversity indices in some secondary cove stands than in old-growth stands when these indices were based on density. When the indices were based on biomass, however, the old-growth stands had higher diversity indices. This indicates that biomass is more equitably distributed among species in old-growth cove forests than in secondary forests dominated by yellow-poplar. Clear dominance

Table 4.–*Overstory composition of second-growth sites*

Species	MADDRON BALD TRAIL				MIDDLE PRONG				CURRY MOUNTAIN			
	RD[1]	RBA[2]	%F[3]	IV[4]	RD	RBA	%F	IV	RD	RBA	%F	IV
Liriodendron tulipifera	42.3	82.9	100.0	55.3	12.2	46.5	100.0	26.1	2.4	15.1	80.0	9.6
Acer rubrum	40.1	12.1	80.6	28.4	9.5	14.1	40.0	10.5	18.3	16.4	86.7	15.7
Halesia tetraptera	4.4	1.6	16.1	4.2	0.6	0.9	6.7	0.9	1.6	2.3	20.0	2.2
Betula lenta	1.8	0.8	9.7	2.2	1.3	0.9	13.3	1.6	1.3	2.3	26.7	2.5
Cornus florida	2.4	0.4	6.5	1.8	10.2	2.3	26.7	5.9	20.0	5.3	40.0	10.3
Robinia pseudoacacia	0.3	0.6	9.7	1.6	1.5	3.8	53.3	5.3	0.4	3.0	40.0	3.0
Acer pensylvanicum	3.0	0.2	3.2	1.5	--	--	--	--	9.7	2.3	20.0	4.9
Tsuga canadensis	2.7	0.4	3.2	1.5	5.7	0.5	6.7	2.5	7.1	3.0	33.3	4.9
Acer saccharum	1.1	0.2	3.2	0.9	22.2	8.0	53.3	13.6	25.2	19.4	93.3	19.3
Quercus rubra	1.1	0.2	3.2	0.9	0.04	0.5	6.7	0.6	1.3	6.6	80.0	6.4
Carpinus caroliniana	0.7	0.2	3.2	0.8	5.7	0.5	6.7	2.5	1.8	0.3	6.7	1.0
Fraxinus americana	0.1	0.2	3.2	0.5	--	--	--	--	--	--	--	--
Nyssa sylvatica	0.1	0.2	3.2	0.5	1.5	1.4	20.0	2.3	1.0	1.0	13.3	1.3
Liquidambar styraciflua	--	--	--	--	17.0	11.3	73.3	14.3	--	--	--	--
Quercus alba	--	--	--	--	3.7	1.9	6.7	2.3	--	--	--	--
Ilex opaca	--	--	--	--	3.2	0.5	6.7	1.7	--	--	--	--
Prunus serotina	--	--	--	--	1.5	0.9	13.3	1.7	--	--	--	--
Carya glabra	--	--	--	--	0.8	0.9	13.3	1.5	1.4	3.3	26.7	2.8
Juglans nigra	--	--	--	--	0.7	0.9	13.3	1.4	--	--	--	--
Carya cordiformis	--	--	--	--	0.5	0.9	13.3	1.4	0.3	1.3	26.7	1.8
Sassafras albidum	--	--	--	--	0.9	0.9	6.7	1.1	1.6	1.6	6.7	1.4
Carya tomentosa	--	--	--	--	0.3	0.9	6.7	0.8	--	--	--	--
Betula alleghaniensis	--	--	--	--	0.4	0.5	6.7	0.7	2.0	0.7	13.3	1.5
Aesculus octandra	--	--	--	--	0.4	0.5	6.7	0.7	--	--	--	--
Quercus montana	--	--	--	--	0.1	0.5	6.7	0.6	2.5	14.1	60.0	8.4
Carya ovata	--	--	--	--	--	--	--	--	0.2	0.7	13.3	0.9
Magnolia fraseri	--	--	--	--	--	--	--	--	1.1	0.7	6.7	0.9
Tilia heterophylla	--	--	--	--	--	--	--	--	1.0	0.3	6.7	0.8
Magnolia acuminata	--	--	--	--	--	--	--	--	0.2	0.3	6.7	0.5

[1] Relative density.
[2] Relative basal area.
[3] Percent frequency.
[4] Importance value.

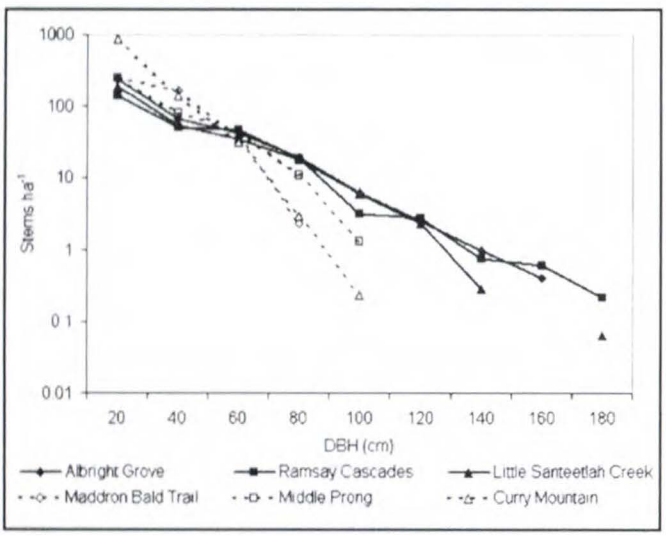

Figure 1.–*Site diameter distributions.*

by any one species did not occur at Curry Mountain, which is reflected by this site's relatively high diversity and evenness indices (table 2).

Regeneration

Regeneration site attributes are presented in table 5. Mean tree species richness of the regeneration component was higher in the secondary sites (mean = 16.3) than in the old-growth sites (mean = 9.7). Mean shrub species richness was higher in the old-growth sites, although numbers and types of shrub species present varied considerably across sites. Mean tree species regeneration density was higher in the secondary sites (mean = 3,979.1 ha-1) than in the old-growth sites (mean = 2,573.7 ha-1). Mean shrub species density was also higher in the secondary sites

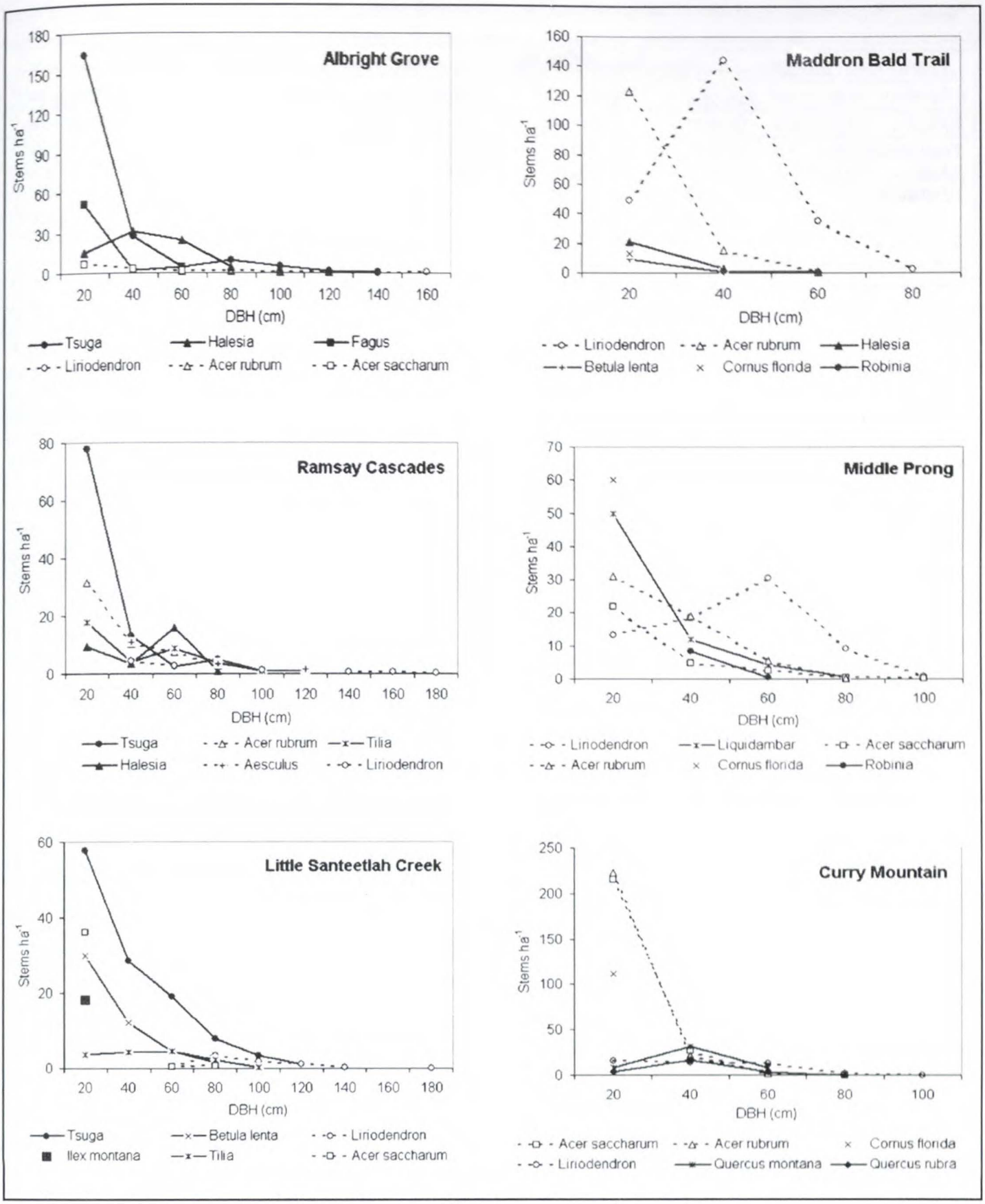

Figure 2.–*Species diameter distributions by site.*

Table 5.–*Regeneration attributes of old-growth and second-growth sites*

	OLD-GROWTH			SECOND-GROWTH		
Site	Albright Grove	Ramsay Cascades	Little Santeetlah Creek	Maddron Bald Trail	Middle Prong	Curry Mountain
Tree density ha-1	2,933.0	2,750.0	2,038.2	4,785.9	5,920.7	1,230.8
Shrub density ha-1	—	1,850.0	1,651.2	490.2	2,000.3	4,830.9
H' (density)	2.40	1.60	3.38	3.36	2.90	2.96
E (density)	0.76	0.62	0.89	0.79	0.70	0.83
D (density)	0.76	0.52	0.86	0.87	0.81	0.82
Tree richness	9	6	14	19	18	12
Shrub richness	1	3	8	4	4	1

(mean = 2,440.5 ha-1) than in the old-growth sites (mean = 1,167.1 ha-1). Mean Shannon-Wiener diversity indices were higher in the secondary sites (mean = 3.1) than in the old-growth sites (mean = 2.5). Equitability indices spanned a greater range in the old-growth sites, but means were similar between the old-growth and secondary sites. Simpson's diversity indices again yielded results identical to the Shannon-Wiener indices.

The composition of the regeneration strata within both old-growth and secondary sites was dominated by a combination of shade tolerant canopy species, understory tree species, and a site-specific assemblage of shrub species. For example, at Albright Grove, the three most important regeneration species were sugar maple, American beech, and eastern hemlock (table 6). Red maple, yellow buckeye, and the shrub species *Viburnum* spp. were the most important species at Ramsay Cascades, the old-growth site with the lowest regeneration diversity. Little Santeetlah Creek was the most diverse old-growth site, and dominance of the regeneration component was relatively evenly distributed between a variety of canopy species, understory tree species, and a species-rich shrub layer dominated by serviceberry (*Amelanchier* spp.).

Red maple, striped maple, and Carolina silverbell had the highest importance values at Maddron Bald Trail, the secondary site with the highest regeneration diversity (table 7). Sweetgum had the second highest importance value in both the overstory and regeneration at Middle Prong. Regeneration at the Curry Mountain site was heavily dominated by the shrub species *Gaylusacia* spp.

Yellow-poplar was a relatively rare regeneration species in the cove forests sampled in this study, with minor importance values at the two sites where it was present, Little Santeetlah Creek and Curry Mountain. Oaks and hickories were common in the regeneration component of the secondary sites. Black locust was present in the regeneration component of two of the secondary sites, and American chestnut sprouts (*Castanea dentata* (Marsh.) Borkh.) were encountered at the Maddron Bald Trail site. Rosebay rhododendron was present in all three old-growth sites and one secondary site. Due to the tendency of rhododendron to exist in discrete patches or thickets commonly associated with riparian zones in this region, its presence in a sub-plot was merely noted and it was not treated similarly to other regeneration-level species in the analyses in this study. The dynamics of rhododendron in the southern Appalachians have been addressed by McGee and Smith (1967), Phillips and Murdy (1985), and Hedman and Van Lear (1995), among others.

Sorensen's similarity indices indicated a greater compositional similarity among old-growth site overstories and second-growth site overstories than between old-growth and second-growth site overstories (table 8). No clear trends were apparent from such comparisons of regeneration at the six study sites. The regeneration component was highly variable between different stands, and this was reflected in the relatively lower regeneration similarity indices. Overall, Sorensen's indices indicated a high degree of similarity between overstory and regeneration composition within individual sites. The lower similarity index for Ramsay Cascades is likely an artifact of the low regeneration species richness at that site.

CONCLUSION

Differences in community attributes were apparent between old-growth and secondary

Table 6.–*Regeneration composition of old-growth sites*

Tree species	ALBRIGHT GROVE			RAMSAY CASCADES			L. SANTEETLAH CREEK		
	RD[1]	% F[2]	IV[3]	RD	% F	IV	RD	% F	IV
Acer saccharum	38.6	50.0	33.6	--	--	--	6.6	22.6	7.6
Fagus grandifolia	25.0	16.7	17.3	--	--	--	2.4	12.9	3.7
Tsuga canadensis	12.5	33.3	15.8	4.3	25.0	8.1	4.5	19.4	5.9
Acer pensylvanicum	9.1	25.0	11.7	3.3	12.5	4.6	10.8	22.6	9.7
Halesia tetraptera	3.4	16.7	6.5	--	--	--	5.6	16.1	5.8
Betula alleghaniensis	8.0	8.3	6.4	1.1	12.5	3.5	--	--	--
Magnolia fraseri	1.1	8.3	3.0	--	--	--	--	--	--
Prunus serotina	1.1	8.3	3.0	--	--	--	--	--	--
Fraxinus americana	1.1	8.3	3.0	--	--	--	1.4	9.7	2.5
Acer rubrum	--	--	--	40.2	50.0	31.9	--	--	--
Aesculus octandra	--	--	--	3.3	37.5	10.5	1.4	6.5	1.9
Nyssa silvatica	--	--	--	7.6	12.5	6.8	--	--	--
Betula lenta	--	--	--	--	--	--	7.0	22.6	7.7
Ilex montana	--	--	--	--	--	--	8.0	9.7	5.9
Ilex opaca	--	--	--	--	--	--	2.1	9.7	2.9
Liriodendron tulipifera	--	--	--	--	--	--	2.4	6.5	2.5
Asimina triloba	--	--	--	--	--	--	1.4	6.5	1.9
Tilia heterophylla	--	--	--	--	--	--	0.7	6.5	1.6
Quercus rubra	--	--	--	--	--	--	0.7	6.5	1.6
Shrub species									
Viburnum spp.	--	--	--	39.1	50.0	31.4	--	--	--
Cornus alternifolia	--	--	--	1.1	12.5	3.5	2.1	12.9	3.5
Amelanchier spp.	--	--	--	--	--	--	22.0	12.9	13.4
Calycanthus floridus	--	--	--	--	--	--	11.9	16.1	9.0
Hamamelis virginiana	--	--	--	--	--	--	3.8	29.0	7.4
Euonymus americanus	--	--	--	--	--	--	1.7	9.7	2.7
Lindera benzoin	--	--	--	--	--	--	2.8	6.5	2.6
Leucothoe fontanesiana	--	--	--	--	--	--	0.3	3.2	0.8
Rhododendron maximum	X[4]	X	X	X	X	X	X	X	X

[1]Relative density.
[2]Percent frequency.
[3]Importance value.
[4]X indicates presence of rhododendron at a site.

sites in this study. Whether or not the compositional differences among secondary site overstories is an artifact of their respective disturbance regimes or other factors is unclear. Yellow-poplar has been noted to grow in almost pure stands on clearcuts and abandoned fields in this region (Buckner and McCracken 1978, Beck 1990). If the Maddron Bald Trail site was not affected by corporate logging (Pyle 1985), evidence suggests that it was an abandoned agricultural site. The diameter distribution at this site also suggests that it originated as an even-aged stand.

The high incidence of dead-standing black locust and sassafras trees witnessed at the Curry Mountain site suggests that the even-aged origin of this stand has been masked by the mortality of a number of pioneer species that comprised the initial overstory. The relatively greater oak-hickory component and steeper slopes at Curry Mountain also indicate that this site may be more accurately characterized as a cove transition forest. This forest type is spatially located between mesic cove forests and more xeric oak-hickory forests as described by Whittaker (1956). The high species richness at Curry Mountain may therefore be related to the co-existence of species common to these two different community types.

Middle Prong is the lowest elevation cove forest in this study, and this is likely related to the importance of sweetgum at this site (Kemp 1993). The diameter distribution at Middle Prong and lack of evidence of self-thinning

Table 7.–*Regeneration composition of second-growth sites*

Tree species	MADDRON BALD TRAIL			MIDDLE PRONG			CURRY MOUNTAIN		
	RD[1]	% F[2]	IV[3]	RD	% F	IV	RD	% F	IV
Acer pensylvanicum	22.0	80.6	20.0	3.0	20.0	3.7	4.1	30.8	9.2
Acer rubrum	15.4	61.3	14.5	4.4	13.3	3.7	1.5	7.7	2.6
Halesia tetraptera	10.0	41.9	9.7	21.5	80.0	19.5	1.0	15.4	4.1
Acer saccharum	8.3	32.3	7.8	8.4	13.3	5.7	--	--	--
Quercus rubra	5.1	38.7	6.9	--	--	--	--	--	--
Betula alleghaniensis	6.6	29.0	6.5	0.3	6.7	0.9	--	--	--
Betula lenta	6.4	25.8	6.1	--	--	--	--	--	--
Tsuga canadensis	5.6	19.4	5.0	0.3	6.7	0.9	0.5	7.7	2.1
Fraxinus americana	2.4	22.6	3.7	0.3	6.7	0.9	--	--	--
Magnolia fraseri	1.2	16.1	2.4	--	--	--	0.5	7.7	2.1
Carpinus caroliniana	3.2	6.5	2.3	6.1	53.3	8.9	1.0	7.7	2.3
Cornus florida	1.2	12.9	2.1	2.7	33.3	5.0	6.6	38.5	12.3
Nyssa silvatica	1.0	6.5	1.2	0.7	13.3	1.8	1.0	15.4	4.1
Sassafras albidum	0.5	6.5	1.0	2.0	20.0	3.2	--	--	--
Magnolia acuminata	0.5	6.5	1.0	--	--	--	--	--	--
Castanea dentata	0.5	3.2	0.6	--	--	--	--	--	--
Magnolia tripetala	0.2	3.2	0.5	--	--	--	--	--	--
Carya glabra	0.2	3.2	0.5	--	--	--	--	--	--
Carya cordiformis	0.2	3.2	0.5	--	--	--	--	--	--
Liquidambar styraciflua	--	--	--	21.5	66.7	18.0	--	--	--
Ilex opaca	--	--	--	0.7	13.3	1.8	--	--	--
Aesculus octandra	--	--	--	1.0	6.7	1.3	--	--	--
Robinia pseudoacacia	--	--	--	0.7	6.7	1.1	0.5	7.7	2.1
Prunus serotina	--	--	--	0.3	6.7	0.9	--	--	--
Quercus alba	--	--	--	0.3	6.7	0.9	--	--	--
Juglans nigra	--	--	--	0.3	6.7	0.9	--	--	--
Pinus strobus	--	--	--	--	--	--	2.5	15.4	4.9
Liriodendron tulipifera	--	--	--	--	--	--	0.5	7.7	2.1
Morus rubra	--	--	--	--	--	--	0.5	7.7	2.1
Shrub species									
Lindera benzoin	6.1	9.7	4.2	11.4	40.0	10.1	--	--	--
Amelanchier spp.	1.7	19.4	3.0	3.7	20.0	4.1	--	--	--
Viburnum spp.	1.5	3.2	1.1	--	--	--	--	--	--
Calycanthus floridus	--	--	--	2.7	6.7	2.1	--	--	--
Gaylusacia spp.	--	--	--	7.4	13.3	5.2	79.7	46.2	50.6
Rhododendron maximum	X[4]	X	X	--	--	--	--	--	--

[1]Relative density.
[2]Percent frequency.
[3]Importance value.
[4]X indicates presence of rhododendron at a site.

suggest a more complex disturbance history consistent with the diffuse disturbance classification of Pyle (1985).

The high species richness and diversity of the secondary sites is related to the presence of less tolerant species such as black locust that presumably became established following the major disturbance affecting those sites. Successional theory suggests that these species will eventually be replaced by more tolerant species, which could lower site species richness over time (Busing 1998). High old-growth diversity and equitability indices based on biomass (Busing 1998, Clebsch and Busing 1989) and basal area reflect that overstory dominance in old-growth mixed mesophytic stands is typically shared by multiple species (Braun 1950, Whittaker 1956).

Table 8.–*Sorensen's similarity indices*

Site	OLD-GROWTH			SECOND-GROWTH		
	Albright Grove	Ramsay Cascades	Little Santeetlah Creek	Maddron Bald Trail	Middle Prong	Curry Mountain
Albright Grove	**0.737**[1]	0.737[2]	0.692	0.435	0.424	0.516
Ramsay Cascades	*0.400*[3]	**0.533**	0.640	0.455	0.438	0.467
Little Santeetlah Creek	*0.522*	*0.300*	**0.733**	0.552	0.513	0.649
Maddron Bald Trail	*0.500*	*0.400*	*0.424*	**0.688**	0.611	0.706
Middle Prong	*0.519*	*0.500*	*0.438*	*0.595*	**0.780**	0.727
Curry Mountain	*0.381*	*0.444*	*0.308*	*0.516*	*0.533*	**0.606**

[1]Bold text: within site comparisons
[2]Plain text: overstory comparisons
[3]Italics: regeneration comparisons

The old-growth and second-growth cove forests sampled in this study suggest a continuity of species composition over successional time. Characteristic cove forest canopy species were typically present in the regeneration layers of these forests with the notable exception of yellow-poplar, which currently showed very little evidence of successful regeneration in any of the study sites. Although the overstory at Maddron Bald Trail was the least diverse of the second-growth sites, possibly reflecting the relative severity of an agricultural-related disturbance history, the regeneration component at this site was the most species rich of all the study sites. As the yellow-poplar overstory begins to thin, other more shade-tolerant cove forest canopy species should be present to replace it, bringing the overstory composition of this site closer to that found in old-growth mixed mesophytic forests.

Continued monitoring of second-growth southern Appalachian cove forests is essential to understanding the ecological processes involved in the succession of these forests towards the composition and structure seen in old-growth cove forests. Continued monitoring of old-growth cove forests is also essential to the establishment of solid baseline data related to these forests. This baseline data is not only useful for comparison with successional sites, but would also be highly useful in the advent of unforeseen future shifts in composition within old-growth cove forests. For example, if the hemlock woolly adelgid (*Adelges tsugae* Annand) eventually reaches the southern Appalachian region with the same impacts seen in some New England states (Jenkins and others 1999), the composition and structure of old-growth cove forests could be dramatically altered. With a solid foundation of baseline data, however, sensitive areas could be identified and impacts could be addressed and possibly mediated.

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EVALUATING MANAGEMENT PRACTICES FOR LOG-GROWN SHIITAKE PRODUCTION IN MIDWESTERN AGROFORESTRY

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ABSTRACT.—Two experiments evaluating outdoor shiitake cultivation practices in the Central United States were established at the same site in December 1999 and May 2000. Natural fruiting began in August 2000. We compare two response variables: traditional Biological Efficiency and W/UV (the weight of harvested mushrooms in g per 100-cm^3 of initial undiscolored wood volume). Use of W/UV provided clearer statistical distinction among experimental factors. Significant differences were detected in year 2000 data among shiitake strains, substrate log species, and spawn forms. We expect shifts in pattern of treatment differences as the experiments proceed.

Values derived from woodland ownership can be enhanced by capitalizing on natural associations of specialty fungi with trees. Mushroom production can be integrated with most management objectives and provides profitable avenues for recycling low-value forestry by-products. A research and demonstration project has been initiated at the University of Missouri Center for Agroforestry (UMCA) to develop and/or identify best management practices for woodland cultivation of specialty mushroom fungi, including shiitake (*Lentinula edodes* (Berk.) Pegler). Although about two-thirds of Central States woodland mushroom growers have other primary sources of income, nearly all consider mushroom cultivation a part of their career, and nearly all grow shiitake (Bruhn and others 2000).

Biological challenges to shiitake cultivation involve selection and development of "best management practices" for improving efficiency of outdoor mushroom production under various sets of local conditions. While a great deal of descriptive literature exists (e.g., Ito 1978, Stamets and Chilton 1983, Chang and Miles 1989, Przybylowicz and Donoghue 1990, Kozak and Krawczyk 1993, Stamets 1993, Ikegaya 1997), published experimental evaluations of outdoor management factors relevant to the Central States region are scarce (e.g., Abe 1989, Bratkovich 1991, Tokimoto and others 1998). We present here an experimental approach capable of distinguishing the effects of field treatments, and first results of two experiments. We also discuss the shortcomings of so-called Biological Efficiency as a response variable for scientific evaluation of log-grown shiitake production, and we suggest use of an alternative response variable, W/UV (the weight of harvested mushrooms in g per 100-cm^3 of initial undiscolored wood volume).

RESEARCH GOAL AND OBJECTIVES

Our long-term research goals are to identify the best management practices for woodland cultivation of shiitake in temperate agroforestry, and to determine the associated productivity.

Specific research objectives of the two experiments presented here are to:

1) compare the productivity of several strains of shiitake;
2) compare the productivity of several readily available host tree species;
3) compare the productivity of the most commonly available forms of spawn (inoculum);
4) evaluate the productivity associated with early winter and spring inoculation;
5) evaluate the relative productivity of immediate vs. delayed inoculation;

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6) compare the results obtained from comparable General Linear Model analyses using two different response variables: traditional "biological efficiency" (BE, the fresh weight of harvested mushrooms expressed as a percentage of initial log dry weight) and W/UV (the fresh weight of harvested mushrooms in g per 100-cm^3 of undiscolored wood volume).

MATERIALS AND METHODS

Study Site and Experimental Design

Two separate experiments were established in early December 1999 and May 2000 at the University of Missouri-Columbia's Horticulture and Agroforestry Research Center (HARC), New Franklin, MO. The experiments are located on a gentle north-facing slope (16 to 24 percent, averaging 21 percent) under hardwood shade.

The experimental units are logs 1-m long and 8- to 23-cm diameter, cut from the stems of healthy, living, pole-size trees. Both experiments included logs cut from sugar maple (*Acer saccharum* Marsh.), white oak (*Quercus alba* L.), and northern red oak (*Q. rubra* L.). All logs were cut from relatively straight stem sections free of heart rot, and with intact bark. Aluminum tags identifying the source tree and stem position were attached to each log at the time of harvest. For example, substrate logs 1-1 and 1-2 are the first and second logs (numbered from the stump upward) cut from tree number 1. These tags are used to relate the mushrooms harvested from each log to that log's descriptive data and the treatments applied to it. For the early winter-initiated experiment (Experiment 1), all logs were cut from trees harvested dormant (10-12 November, after passage of autumn color). Inoculation was conducted 8-10 December. In the spring-initiated experiment (Experiment 2), logs cut from trees harvested dormant in mid-February (approximately 13-weeks prior to inoculation) are compared with logs cut from trees harvested 2-5 May. All logs were inoculated in May.

The surface area (SA) of each substrate log was calculated from its length and diameters outside bark at both ends. Undiscolored wood volume was calculated for each log from its length and the diameters inside bark and inside undiscolored wood at both ends. The numbers and diameters of live branch stubs on each log were also recorded. The initial dry weight of each log was estimated from it's initial fresh weight and the fresh-to-dry-weight ratio of a 1-cm thick slice collected from the end of each log at the time of harvest; for example, a single slice taken between logs 1-1 and 1-2 at the time of felling and bucking was used to represent the initial fresh-to-dry-weight ratio of both logs.

The same three fungal strains are evaluated in both experiments. All inoculum (spawn) was purchased from Field and Forest Products, Peshtigo, WI. Strains included in both experiments represent one each selected for its fruiting potential during cold weather (CW-50), warm weather (WW-44), or over a wide temperature range (WR-46).

Two spawn forms of each shiitake strain are being evaluated in each experiment. Standard sawdust and wooden dowel spawn (Przybylowicz and Donoghue 1990, Kozak and Krawczyk 1993), both sealed with hot cheese wax, are being compared in Experiment 1. Standard sawdust (sealed with hot cheese wax) and waxless "thimble" spawn (molded sawdust-based plugs with a thin styrofoam cap) are being compared in Experiment 2.

Logs were inoculated as follows. For all spawn forms, holes were drilled approximately 15-cm apart in rows 5-cm apart around the log circumference with a Makita high speed drill (8,000 rpm). A diamond pattern was created by offsetting the holes in alternate rows by approximately 7.5-cm. Holes for sawdust and thimble spawns vs. dowel spawn placement were drilled 2.5-cm deep and 12-mm in diameter vs. 1.9-cm deep and 8.5-mm in diameter, respectively, using screw-tip bits. The screw tip guides the drill bit safely, and the high speed of the drill strips the wood around the screw tip when the bit reaches the pre-set stop depth, permitting very fast hole creation and easy drill extraction (Przybylowicz and Donoghue 1990, Kozak and Krawczyk 1993). Very hot (smoking) molten cheese wax was used to seal traditional sawdust and dowel inocula. If the wax is not hot enough to sizzle on contact with the inoculum, it may not provide a lasting seal over the spawn (Przybylowicz and Donoghue 1990, Kozak and Krawczyk 1993). The styrofoam cap on thimble spawn is designed to provide an effective seal without application of cheese wax.

Each experiment is arranged as a randomized complete block design. Experiments 1 and 2 comprise 9 and 8 blocks, respectively, with one log of each species assigned randomly to a treatment combination and a location within each block. Logs were laid approximately 10-cm above the ground parallel to the forest floor,

oriented perpendicular to the slope contour with centers 30-cm apart, on paired wooden rails placed parallel to the slope contour. Rails were laid over black perforated plastic Weed Block® landscape fabric (Easy Gardener, Inc., Waco, TX). This landscape fabric has functioned well to reduce weed growth among the substrate logs. Leaf and woody litter were removed from the fabric when it contacted the substrate logs. The forest floor around the experimental blocks was mowed as necessary to a height of approximately 5-cm with a heavy-duty brush mower, to further prevent forest floor vegetation from overgrowing the substrate logs.

Air temperature at the study site is being measured 66-cm above ground level, using a Campbell Scientific, Inc. (Logan, UT) micrologger with sensors. Degree days are calculated on a 4° C basis from the average of daily minimum and maximum temperatures. Daily total precipitation is monitored in the open at the HARC weather station, approximately 400-m from the experimental site.

Harvesting and Productivity

The weight and grade of each mushroom harvested is being credited to its source log. Harvest frequency (as often as daily) is intended to maximize grade yield. Individual mushrooms were harvested by twisting and or bending the base of the stem to separate the mushroom stem from the log at or just below the log surface. It is hypothesized that mushrooms harvested in this manner remain fresher longer, and that this method leaves less debris on the log to attract pests (Joe Krawczyk, Field & Forest Products, Inc., personal communication).

Individual mushrooms were assigned to grades 1, 2, or cull (Kozak and Krawczyk 1990). To summarize, Grade 1 mushrooms have approximately round, complete caps 2.5- to 10-cm in diameter, 50–80 percent expanded, less than 10 percent pitting, but free of rodent damage. Gills are white, unbruised, and clean. Grade 1 mushrooms are neither excessively dry nor water-soaked. A Grade 2 mushroom is any usable mushroom which does not meet specifications for Grade 1, and a Cull mushroom is any unusable mushroom. The distinction between Grade 2 and Cull mushrooms varies, depending on the end uses available to a given grower. The analyses presented here are based on total fresh weight of harvested mushrooms, regardless of grade.

Models

All statistical analyses were conducted using SAS/STAT System Release 8.0 (SAS Inst., Inc., Cary, NC, USA). SAS PROC CORR was used to evaluate the extent of correlation among harvest-related and log characteristic variables. SAS PROC GLM was used to analyze the effects of treatments and log characteristics on productivity in each experiment during the year 2000. Two response variables were evaluated: the traditional "biological efficiency" (BE, the fresh weight of mushrooms produced from a substrate log expressed as a percentage of initial log dry weight) and W/UV (the fresh weight of mushrooms produced per 100-cm^3 of undiscolored wood volume). Both of these response variables were calculated from total harvested mushroom weight. Crop grade distribution will be analyzed after collection of another year's data. A third model used W/UV as the response variable, but included substrate log SA as a covariate. Tukey's HSD test ($\alpha = 0.05$) was used to evaluate treatments where ANOVA detected significant differences (Dowdy and Wearden 1983).

RESULTS

Fruiting commenced naturally in Experiments 1 and 2 in early and late August 2000, respectively, and continued into November 2000 until approximately day of year (DOY) 300 (fig. 1). In Experiment 1, strain WR46 began fruiting in response to August rains (after DOY 220), whereas strains CW50 and WW44 fruited little until the rains of October 2000 (after DOY 280, figs. 1 and 2). In Experiment 2, WR46 fruited only weakly in late September 2000, and all three strains responded to October rains with increased fruiting (figs. 1 and 2). Thus, with early winter inoculation, WR46 appeared to complete spawn run earlier than Strains CW50 and WW44, whereas this difference was much less pronounced following late spring inoculation.

During 2001, fruiting commenced naturally in both experiments in April (after DOY 90) and continued into late December (data are shown for only March through July in fig. 3). However, in both experiments, CW50 fruited substantially in April 2001, whereas WR46 and WW44 began fruiting substantially in May 2001 (fig. 3). Fruiting in early 2001 appeared to be more responsive to accumulated degree days than to precipitation events (figs. 3 and 4).

At the end of the first growing season, W, BE, and W/UV for each log were all strongly and positively correlated with one another (table 1).

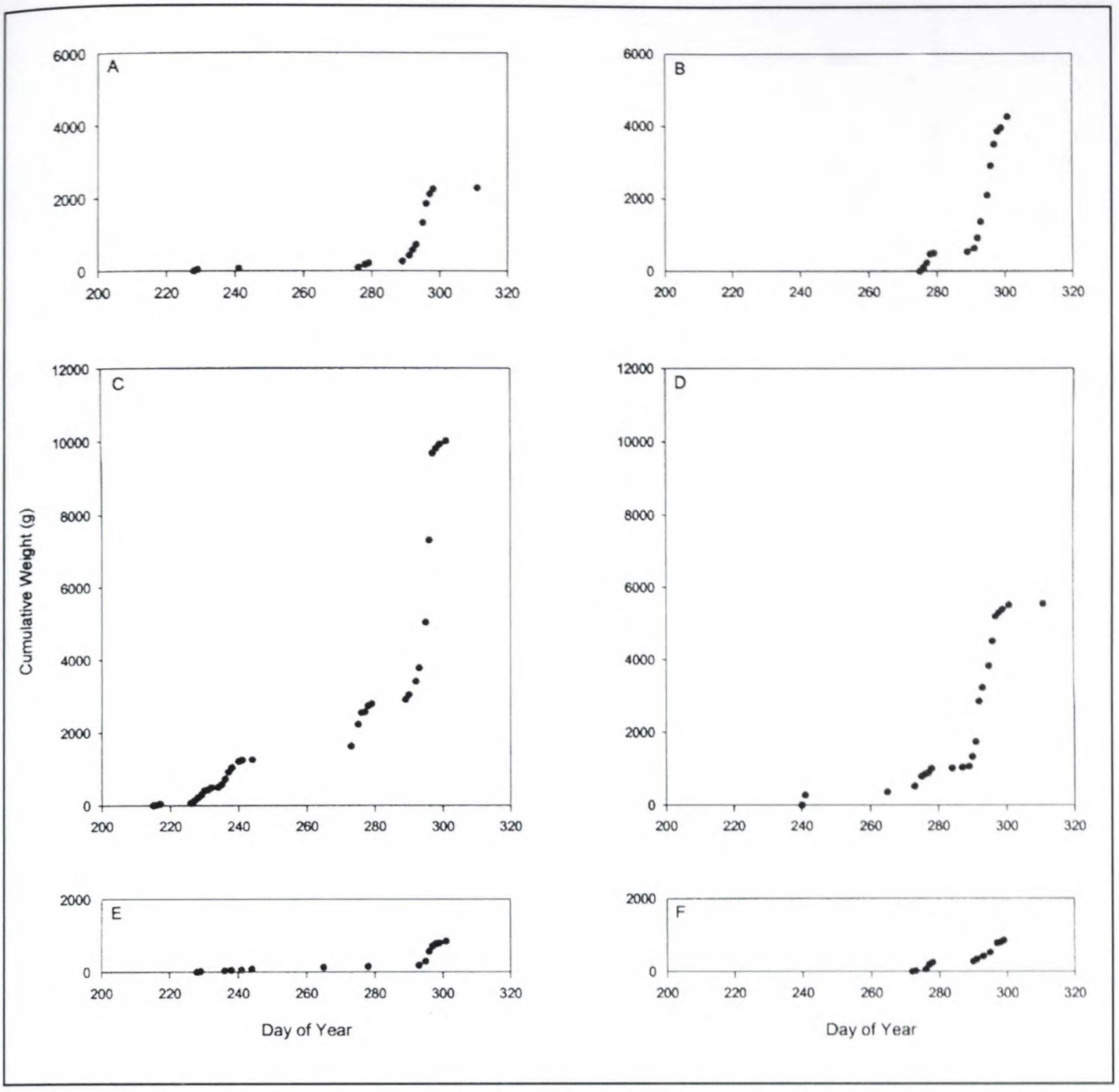

Figure 1.—*Shiitake production during year 2000 by Cold Weather, Wide Range, and Warm Weather shiitake strains inoculated during early December 1999 (panels A, C, and E, respectively) and by the same strains inoculated during May 2000 (panels B, D, and F, respectively).*

In both experiments, W was not correlated with substrate log SA, whereas BE and W/UV were both negatively correlated with SA. W was weakly correlated (Experiment 1) or not correlated (Experiment 2) with UV, whereas BE and W/UV were not correlated and negatively correlated, respectively, with UV in both experiments. Neither W nor W/UV was correlated with the number or total diameter of live branches severed from substrate logs, and BE was only weakly correlated with total severed branch diameter. Substrate log SA and UV are strongly and positively correlated with each other in both experiments, and both are negatively correlated with substrate log branchiness.

ANOVA model results for Experiments 1 and 2 using BE as the response variable are presented

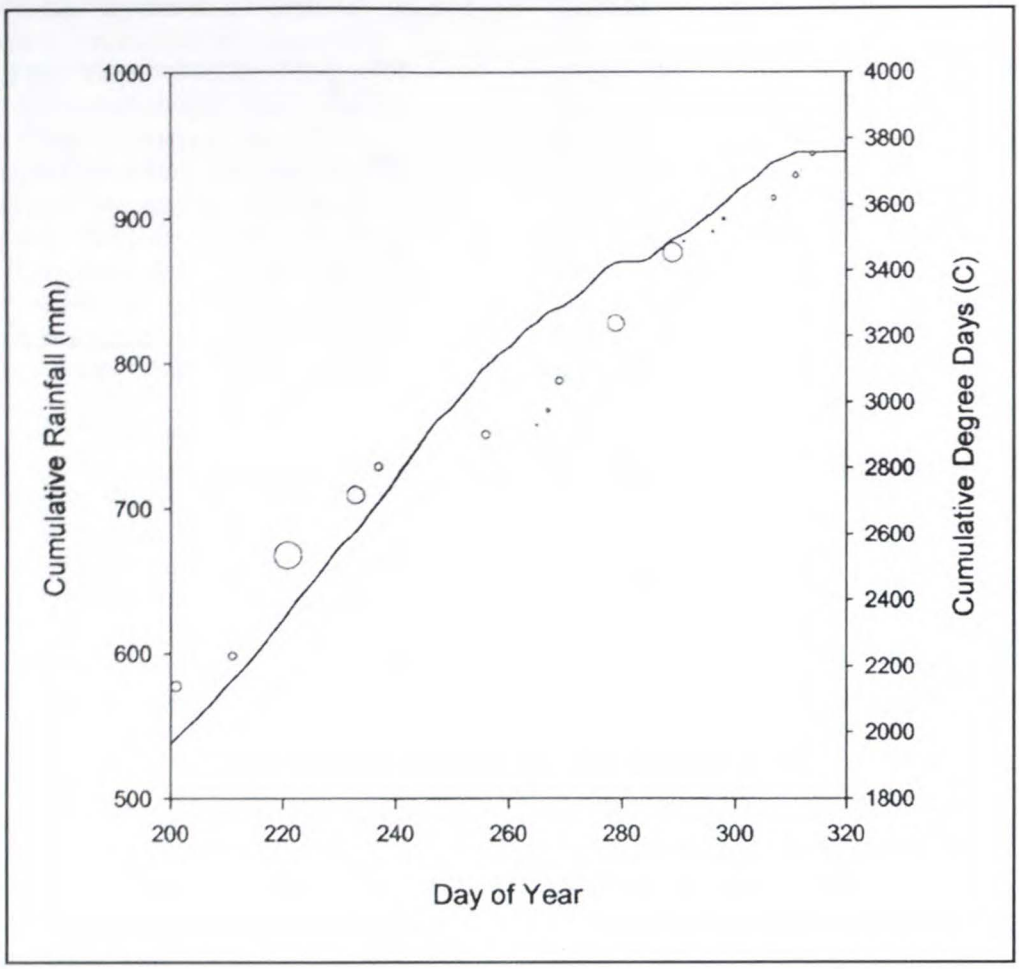

Figure 2.—*Cumulative precipitation (each circle represents a precipitation event, for which circle diameter is proportionate to precipitation amount) and cumulative degree days (solid line, 4º C basis) during year 2000.*

in Appendix table 1. Corresponding model results using W/UV as the response variable are presented in Appendix table 2. The corresponding models presented in Appendix table 3 with W/UV as the response variable include substrate log SA as a covariate. In each experiment, all three models detected the same highly significant differences in productivity among shiitake strains (table 2). In both experiments, WR46 outproduced both CW50 and WW44 (fig. 1). In Experiment 2, CW50 also outproduced WW44. The additional resolution among the three strains achieved in Experiment 2 is probably due to the greater variation in Experiment 1 productivity resulting from the significantly different performance of the two spawn forms employed in Experiment 1 (see below and table 2). A similar pattern appears to have developed through July 2001 (fig. 3).

Differences in productivity among substrate log species were not as pronounced in Experiment 1 as in Experiment 2 (table 2), again probably due to the greater variation in Experiment 1 productivity resulting from the significantly different performance of the two spawn forms employed. Because BE vs. W/UV express productivity as functions of total substrate log weight vs. undiscolored wood volume, it isn't surprising that sugar maple BE exceeded that of white oak (though not significantly greater than red oak) for year 2000 in Experiment 1. Nor is it surprising that red oak W/UV exceeded that of sugar maple (though not significantly greater than white oak) for year 2000 in Experiment 1. However, in Experiment 2, both BE and W/UV for white oak exceeded values for sugar maple during year 2000.

During year 2000, sawdust spawn outperformed dowel spawn in Experiment 1 (table 2). Waxless "thimble" spawn performed as well as traditional sawdust spawn during year 2000 in Experiment 2. The distinction between the

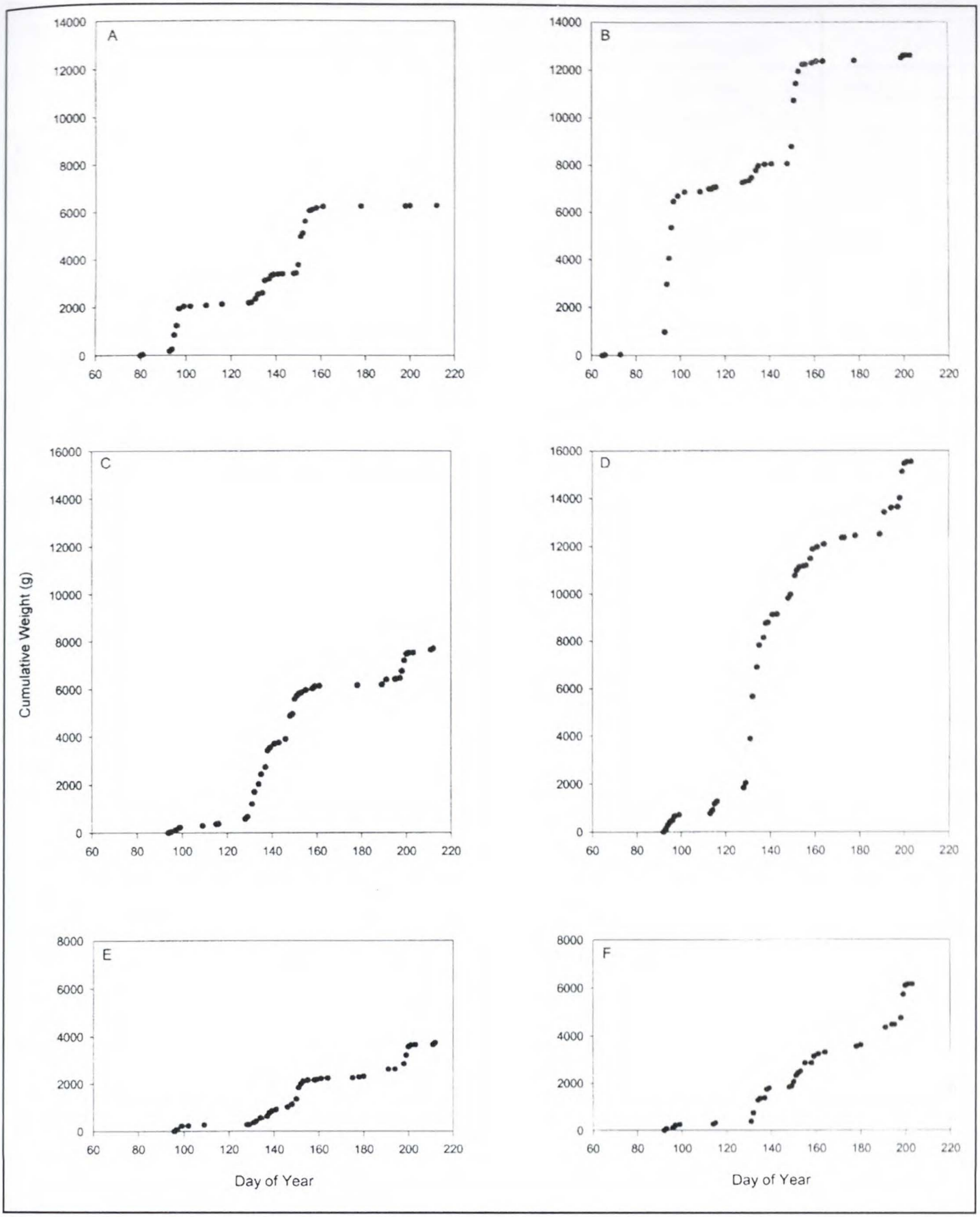

Figure 3.—*Shiitake production during year 2001 (through July) by Cold Weather, Wide Range, and Warm Weather shiitake strains inoculated during early December 1999 (panels A, C, and E, respectively) and by the same strains inoculated during May 2000 (panels B, D, and F, respectively).*

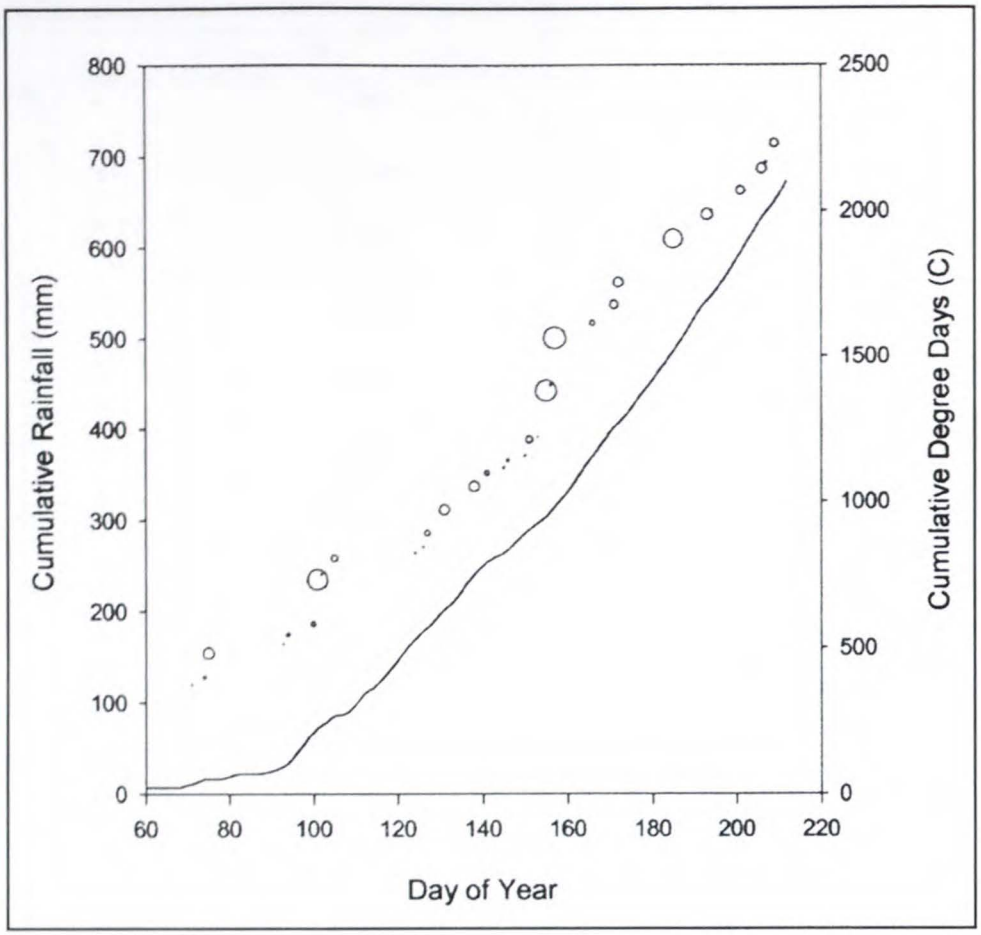

Figure 4.—*Cumulative precipitation (each circle represents a precipitation event, for which circle diameter is proportionate to precipitation amount) and cumulative degree days (solid line, 4º C basis) during year 2001 through July.*

performance of sawdust and dowel spawn in Experiment 1 was stronger when productivity was measured as W/UV than as BE.

Delayed inoculation (13 weeks following harvest) had no effect on productivity during year 2000 measured either as BE or as W/UV (table 2). A significant block effect was detected for year 2000 fruiting in Experiment 2 only, and only for W/UV. Of the eight blocks, block 3 was more productive than blocks 4 and 7.

A stronger interaction between shiitake strain and substrate log species was detected in both experiments when productivity was measured as W/UV rather than as BE (table 2). A significant interaction between shiitake strain and spawn form was detected only in Experiment 1 and only for W/UV. A substrate log species interaction with inoculation delay was detected in Experiment 2 only for BE.

DISCUSSION

Our studies of outdoor log-grown shiitake production are designed to contribute to development of best management practices for growers in the Central States. The experiments presented here have an expected duration of 4 years, through year 2003. In the course of evaluating BE and W/UV, and comparing their relationships with experimental factors throughout the life of these two experiments, we expect to see some shifts in the relative productivity associated with the experimental factors (i.e., shiitake strain, substrate species, spawn form, and inoculation delay), including development of significant statistical interactions among factors. While it is premature to draw conclusions based solely on the first season's data, the novel analytical approach described here will provide improved insight into the biology and management of log-grown shiitake mushroom production.

Table 1.–*Correlation coefficients R^2 and significance levels for two measures of mushroom productivity with selected substrate log characteristics for 2000 (first field season) in each of two experiments*[1]

	W[3]	SA[4]	UV[5]	W/UV[6]	N of stubs[7]	Stub diam.[8]
			Experiment 1 (inoculated early December 1999)			
BE[2]	0.848 ***	-0.296 ***	-0.020 ns	0.825 ***	0.068 ns	0.087 ns
W		-0.059 ns	0.193 *	0.780 ***	-0.033 ns	-0.025 ns
SA			0.503 ***	-0.239 **	-0.428 ***	-0.391 ***
UV				-0.172 *	-0.348 ***	-0.303 ***
W/UV					0.222 ns	0.110 ns
			Experiment 2 (inoculated May 2000)			
BE	0.864 ***	-0.204 ***	-0.096 ns	0.868 ***	0.109 ns	0.121 *
W		0.073 ns	0.081 ns	0.795 ***	0.068 ns	0.096 ns
SA			0.598 ***	-0.148 *	-0.288 ***	-0.240 ***
UV				-0.231 ***	-0.183 **	-0.129 *
W/UV					0.109 ns	0.102 ns

[1]Tabular symbols indicate significance at $P \leq 0.05$ (*), $P \leq 0.01$ (**), $P \leq 0.001$ (***), not significant (ns).
[2]BE, Biological Efficiency, the fresh weight of all mushrooms harvested from a substrate log expressed as a percentage of initial log dry weight.
[3]W, Weight, the total fresh weight (g) of shiitake mushrooms harvested from each substrate log.
[4]SA, Surface Area, the surface area (cm^3, outside bark) of each substrate log.
[5]UV, Undiscolored wood volume, the volume of undiscolored wood (cm^3) in each substrate log, estimated from diameter inside bark and undiscolored wood thickness at both ends of each log.
[6]W/UV, the weight of harvested mushrooms (g) per 100-cm^3 of initial undiscolored wood volume.
[7]Number of Stubs, the number of branch stubs alive at the time of harvest on each substrate log.
[8]Stub Diameter, the total diameter (cm) of all branch stubs alive at the time of harvest on each substrate log.

Table 2.–*Patterns of significance detected by three ANOVA models of shiitake production during 2000 applied to two experiments*

	EXPERIMENT 1[2]			EXPERIMENT 2[2]		
Source of variation [1]	**BE None**	**W/UV None**	**W/UV SA**	**BE None**	**W/UV None**	**W/UV SA**
Shiitake Strain (SS)	*** [3]	***	***	***	***	***
Log Species (LS)	*	*	**	***	***	***
SS x LS	*	***	***	**	***	***
Inoculum Form (IF)	*	***	***	ns	ns	ns
SS x IF	ns	**	***	ns	ns	ns
LS x IF	ns	ns	ns	ns	ns	ns
Inoculation Delay (ID)	—	—	—	ns	ns	ns
SS x ID	—	—	—	ns	ns	ns
LS x ID	—	—	—	*	ns	ns
IF x ID	—	—	—	ns	ns	ns
Block	ns	ns	ns	ns	*	*
Log Surface Area	—	—	***	—	—	*

[1]Variables included: Shiitake Strain (3 - Cold Weather, Wide Range, Warm Weather); Log Species (3 - *Acer sacharum, Quercus alba, Q. rubra*); Inoculum Form (2 - sawdust and dowel inocula used December 1999, sawdust and thimble inocula used May 2000); Inoculation Delay (2 - only for May 2000 inoculation, no delay or 4-mo delay post-harvest); Block (9 for December 1999 experiment, 8 for May 2000 experiment); Surface Area (the surface area outside bark of each substrate log, cm^3).
[2]Experiments 1 and 2, respectively, represent logs inoculated either in early December 1999 or May 2000. Model response variables were either Biological Efficiency (BE, the fresh weight of all mushrooms harvested from a substrate log expressed as a percentage of initial log dry weight) or Weight/Undiscolored wood volume (W/UV, total weight of harvested mushrooms (g) per 100-cm^3 of undiscolored wood volume for individual shiitake logs. Log Surface Area (SA) was either included in the model as a covariate or not (None).
[3]Tabular symbols indicate significance of F: *, $P \leq 0.05$; **, $P \leq 0.01$; ***, $P \leq 0.001$; ns, not significant; —, not tested.

We were motivated to compare BE and W/UV as alternative response variables for modeling log-based shiitake mushroom productivity for two reasons. First, differences between red and white oaks and maples in stemwood anatomy and heartwood composition may result in different rates and patterns of substrate log decomposition by shiitake mycelium, resulting in different efficiencies of wood energy conversion into mushroom biomass. Certain tree genera (including oaks) routinely produce highly discolored heartwood through the deposition of toxic extractives in dying sapwood cells. These toxic extractives are responsible for the natural decay resistance of heartwood (Zabel and Morrell 1992). Other tree genera (including maples) normally produce much less decay-resistant heartwood, which is undiscolored due to its lower extractive content. Maple heartwood darkens only through the living tree's compartmentalization response to stem injury (Shigo 1984) and microbial colonization (Shigo 1972). The heartwoods of white oaks, red oaks, and maples have been rated resistant, moderately susceptible, and susceptible to decay, respectively (Scheffer and Cowling 1966).

For the above reasons, we hypothesize that a larger percentage of the volume of maple logs compared to oak logs may be more efficiently utilized by shiitake. If this is the case, then BE (which is based on total log weight) may provide incomplete insight into differences in productivity between oaks and maples. Through consideration of both BE and W/UV as response variables, we expect to both document and explain differences in shiitake productivity between sugar maple and red and white oak. To the extent that undiscolored heartwood and sapwood are better energy sources for shiitake growth than discolored heartwood, W/UV should provide additional useful insight for substrate species comparisons.

Second, the proportion of decay-resistant discolored heartwood in oak logs generally increases as log diameter increases. This phenomenon biases attempts to compare the effects of treatments if they are applied to logs of different sizes. Here again, W/UV may provide useful insight into the effects of log size as well as management practices.

Another interesting substrate log characteristic under study is surface area (SA), which directly measures the space available for mushroom production. Substrate log SA also indirectly reflects the tendency of a log to dry out between wetting events. Though SA was not correlated with W during year 2000 (table 1), SA was significantly correlated with substrate log productivity in year 2000 analyses of W/UV (table 2). We will continue to evaluate the relationship between SA and productivity. The information gained should help us to determine the effect of substrate log size on productivity.

The poor correlations for year 2000 shiitake harvest (W) with either SA or UV probably reflect the early stages of these experiments. We anticipate that these correlations will become significant as the experiments progress and the strains more completely utilize the substrate logs. We are evaluating the correlations between shiitake productivity and substrate log branchiness. Severed branches represent both points of potential invasion by competitor fungi and locations from which excessive substrate drying may take place. As might be expected this early in an experiment, there was no apparent effect of branchiness in the year 2000 analysis.

Productivity by the warm weather strain WW44 in year 2000 of both our experiments clearly lagged behind that of the other two strains, and this trend appeared to continue less markedly through July 2001. Jo Krawczyk (Field & Forest Products, Inc., personal communication) indicated that productivity by WW44 increases more gradually compared with other WW46 and CW50, and that WW44 tends to continue to fruit for a year or two longer than the other strains. We will be able to determine if this is the case under conditions common for central Missouri. Also, the years 2000 and 2001 growing seasons were not particularly warm, and it would be useful to document the performance of WW44 during a warmer growing season. Strain CW50 is performing well in these experiments. This strain fruited substantially in early April 2001 (approximately DOY 93, as much as a month earlier than WR46 or WW44). In fact, some form of protection from spring frost and desiccation prior to overstory foliation would have enhanced yields of all three strains during March and April 2001, when many mushroom primordia were lost to frost and dry weather.

We expect that the substrate species effect on productivity will become clearer as the experiments mature. Because pole-size sugar maple stems characteristically contain less discolored heartwood than do oak stems, one might expect shiitake to decompose sugar maple logs more completely, leading to higher BE values for maple logs than for oak logs. That seems to be

the case already in Experiment 1. Nevertheless, because it may take shiitake mycelium longer to penetrate deeply enough to access all the non-discolored wood in maple logs, it isn't surprising that W/UV appears to be greater for oak than for maple initially. If maple logs are consumed faster and/or more completely due to their less decay-resistant heartwood, use of sugar maple may provide growers a faster and/or greater return on their investment.

The only difference in productivity during year 2000 attributable to spawn form developed in Experiment 1, where traditional sawdust-based spawn outproduced solid wood (dowel) spawn. Solid wood spawn has been recommended for autumn inoculations in colder climates where winter temperatures might result in frost-heaving of more traditional sawdust-based spawn (Kozak and Krawczyk 1993). While we have noticed some winter disturbance of sawdust spawn, the disturbance does not resemble frost heave. Rather, we attribute this disturbance to birds because disturbances are most often conical gouges in the spawn. Six species of woodpeckers have been observed in the study area. No difference developed during year 2000 between the performances of traditional sawdust and thimble spawn.

No difference in productivity has yet developed between logs harvested dormant and inoculated 13-weeks later compared with logs harvested just after bud-break and inoculated immediately. We are concerned that the process of aging cut logs might favor development of *Hypoxylon atropunctatum* and/or other competing wood decay fungi. Although competing wood decay fungi have not been a serious problem yet, we will evaluate fruiting by competing fungi on each log to serve as a covariate to model the impact of competition on shiitake production.

ACKNOWLEDGMENT

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APPENDICES

Appendix Table 1.–*Results of ANOVA using Biological Efficiency of individual shiitake logs in harvest year 2000 as the response variable to detect significant differences among treatments for Experiment 1 (initiated early December 1999) and for Experiment 2 (initiated May 2000)*[1]

Source of variation	df	SS	F-value	Significance of F
EXPERIMENT 1:				
Model	21	71.932	4.75	<0.0001
Shiitake Strain (SS)	[2]	41.877	29.06	<0.0001
Log Species (LS)	[2]	4.669	3.24	0.0422
SS x LS	[4]	9.084	3.15	0.0163
Inoculum Form (IF)	[1]	3.882	5.39	0.0218
SS x IF	[2]	3.963	2.75	0.0675
LS x IF	[2]	0.730	0.51	0.6038
Block	[8]	7.838	1.36	0.2197
Error	135	97.258		
Total	156	169.190		
EXPERIMENT 2 :				
Model	26	63.898	3.96	<0.0001
Shiitake Strain (SS)	[2]	29.334	23.62	<0.0001
Log Species (LS)	[2]	9.966	8.02	0.0004
SS x LS	[4]	9.755	3.93	0.0042
Inoculum Form (IF)	[1]	0.068	0.11	0.7412
SS x IF	[2]	2.895	2.33	0.0995
LS x IF	[2]	0.701	0.56	0.5696
Inoculation Delay (ID)	[1]	2.249	3.62	0.0583
SS x ID	[2]	1.236	1.00	0.3712
LS x ID	[2]	4.207	3.39	0.0355
IF x ID	[1]	0.238	0.38	0.5368
Block	[7]	3.890	0.89	0.5113
Error	236	146.579		
Total	262	210.477		

[1]Biological Efficiency (BE), the fresh weight of all mushrooms harvested from a substrate log expressed as a percentage of initial log dry weight.

Appendix Table 2.–*Results of ANOVA using Weight/Undiscolored wood volume for individual shiitake logs for year 2000 harvest as the response variable to detect significant differences among treatments for Experiment 1 (initiated early December 1999) and Experiment 2 (initiated May 2000)*[1]

Source of variation	df	SS	F-value	Significance of F
EXPERIMENT 1:				
Model	21	582.952	9.08	<0.0001
Shiitake Strain (SS)	[2]	332.392	54.33	<0.0001
Log Species (LS)	[2]	19.149	3.13	0.0469
SS x LS	[4]	89.492	7.31	<0.0001
Inoculum Form (IF)	[1]	41.230	13.48	0.0003
SS x IF	[2]	33.872	5.54	0.0049
LS x IF	[2]	7.844	1.28	0.2808
Block	[8]	45.886	1.87	0.0690
Error	135	412.944		
Total	156	995.896		
EXPERIMENT 2:				
Model	26	109.405	5.33	<0.0001
Shiitake Strain (SS)	[2]	41.848	26.51	<0.0001
Log Species (LS)	[2]	22.908	14.51	<0.0001
SS x LS	[4]	16.570	5.25	0.0005
Inoculum Form (IF)	[1]	0.093	0.12	0.7322
SS x IF	[2]	3.559	2.25	0.1072
LS x IF	[2]	1.869	1.18	0.3078
Inoculation Delay (ID)	[1]	1.338	1.69	0.1942
SS x ID	[2]	1.341	0.85	0.4289
LS x ID	[2]	2.118	1.34	0.2634
IF x ID	[1]	1.649	2.09	0.1496
Block	[7]	14.005	2.53	0.0156
Error	235	185.496		
Total	261	294.902		

[1]Weight/Undiscolored wood volume, total weight of harvested mushrooms (g) per 100-cm^3 of undiscolored wood volume for individual shiitake logs.

Appendix Table 3.–*Results of ANOVA using Weight/Undiscolored wood volume for individual shiitake logs for year 2000 harvest as the response variable (and log surface area as a covariate) to detect significant differences among treatments for Experiment 1 (initiated early December 1999) and Experiment 2 (initiated May 2000)*[1]

Source of variation	df	SS	F-value	Significance of F
EXPERIMENT 1:				
Model	22	615.326	9.85	<0.0001
Shiitake Strain (SS)	[2]	308.822	54.37	<0.0001
Log Species (LS)	[2]	30.063	5.29	0.0061
SS x LS	[4]	75.145	6.61	<0.0001
Inoculum Form (IF)	[1]	38.765	13.65	0.0003
SS x IF	[2]	41.592	7.32	0.0010
LS x IF	[2]	6.857	1.21	0.3023
Block	[8]	38.943	1.71	0.1006
Surface Area	[1]	32.374	11.40	0.0010
Error	134	380.571		
Total	156	995.896		
EXPERIMENT 2:				
Model	27	113.913	5.45	<0.0001
Shiitake Strain (SS)	[2]	41.160	26.61	<0.0001
Log Species (LS)	[2]	20.548	13.28	<0.0001
SS x LS	[4]	15.582	5.04	0.0007
Inoculum Form (IF)	[1]	0.161	0.21	0.6484
SS x IF	[2]	3.322	2.15	0.1191
LS x IF	[2]	2.849	1.84	0.1609
Inoculation Delay (ID)	[1]	2.451	3.17	0.0763
SS x ID	[2]	1.146	0.74	0.4779
LS x ID	[2]	1.572	1.02	0.3636
IF x ID	[1]	1.388	1.79	0.1817
Block	[7]	13.830	2.55	0.0149
Surface Area	[1]	4.507	5.83	0.0165
Error	234	180.989		
Total	261	294.902		

[1]Weight/Undiscolored wood volume, total weight of harvested mushrooms (g) per 100-cm^3 of undiscolored wood volume for individual shiitake logs; surface area, the surface area (cm^3, outside bark) of each substrate log.

INTEGRATING FUEL AND FOREST MANAGEMENT: DEVELOPING PRESCRIPTIONS FOR THE CENTRAL HARDWOOD REGION

Edward F. Loewenstein, Keith W. Grabner, George W. Hartman, and Erin R. McMurry[1]

ABSTRACT.—The oak dominated forests in the Ozarks of southern Missouri evolved under the influence of fire for thousands of years. However, fire exclusion and timber harvests have changed historical fuel loads and modified vegetative structure. The resurgent interest in restoration of fire dependent ecosystems in conjunction with the needs of resource managers to control fuel loads and affect the incidence and severity of wildfire suggests that the use of prescribed fire as a silvicultural tool will continue to increase above current levels. In order to develop the predictive models necessary to safely manage this tool, a replicated study was initiated in 2001. The rationale for this research and results from the pretreatment vegetation survey are presented.

General characteristics of pre-settlement Ozark forests were open woodlands with widely scattered trees and a diverse understory composed of grasses and wildflowers. Journal entries from early explorers in the late 1600s and throughout the 1700s describe the lands surrounding the Ozarks as open, thinly scattered with trees, and grassy (Beilmann and Brenner 1951). The character of the Ozarks was seemingly unchanged through the early 1800s when Henry Rowe Schoolcraft traveled through the area and described the uplands he encountered as open with scattered trees and the ground covered with grasses (Schoolcraft 1821). Observations of other travelers and settlers to this region further support these observations, as do surveys contracted by the General Land Office (GLO) from 1815 to the 1850s (Schroeder 1981).

The journals that provide a glimpse of the historic Ozark vegetation also provided an explanation for the cause of their open character. They described the practice of Native American burning, especially in the fall, as the reason for the open character of the forests (Beilmann and Brenner 1951, Ladd 1991, Nigh 1992). A pyrotechnic history for Central Hardwood forests can also be derived from historic forest structure and the ongoing species composition changes that are occurring in this region (Hicks 1998). A final clue to the fire history of the central hardwoods is found in dendrochronological fire histories. Pre-European settlement fire histories for the Ozarks have been developed that indicate fire return intervals from 2.8 to 12.4 years (table 1) with intervals ranging from 1 to 57 years (Guyette and Cutter 1991, 1997; Guyette 1994; Cutter and Guyette 1994). It is important to remember that dendrochronological fire histories tend to underestimate fire frequencies because not all trees are scared by fires, due to variability in fuels and fire intensities (Guyette and Cutter 1997).

The disturbance regimes responsible for creating the early structure and composition of the Missouri Ozarks changed markedly starting with region wide forest clearing during the Ozark timber boom from 1887 to 1920. After the land was cleared of merchantable trees, many unemployed mill workers tried to make a living by crop farming and raising livestock. Annual burning was used to control the vigorous growth of hardwood sprouts on these farms. The grazing and frequent burning in conjunction with the almost complete removal

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Table 1.—*Pre-European settlement fire histories for the Ozarks*

Area	Mean fire return interval & (range) in years		
	1581 – 1700	1701 - 1820	1821 – 1940
Peck Ranch Conservation Area, Shannon and Carter Counties, Missouri		6.3 (2-24)	3.1 (1-10)
Current River Watershed, Missouri	17.7 (1-60)	12.4 (1-57)	3.7 (1-33)
Huckleberry Hollow, Shannon County, Missouri		7.1 (1-24)	2.2 (1-11)
Turkey Mountain, Marion County, Arkansas		5.7 (1-34), 1770 - 1993	
Laclede County, Missouri		2.8 years, 1740 to 1850 24.0 years, 1851 to 1993	
Caney Mountain Conservation Area, Missouri		4.3 years, 1710 to 1810 6.4 years, 1811 to 1990	

of a pine seed source greatly impacted pine reproduction. When many of these farms were abandoned, the forests regenerated primarily from oak sprouts; the pine component was greatly reduced from pre-exploitation levels (Cunningham and Hauser 1989).

About this same time, fire suppression by the USDA Forest Service and Missouri Department of Conservation became quite effective, nearly eliminating widespread wildfire from the area (Guyette and Cutter 1997). Without recurring fire to hold it in check, the density of the newly regenerating forest was much greater than historic levels. As this new forest matured, the deep shade beneath its closed canopy affected the composition and development of the understory vegetation as well. Thus, primarily because of an altered disturbance regime, we now find four major differences between current and historic forest conditions in the Ozarks:

1) tree density has increased since pre-settlement,
2) a midstory and shrub layer has developed,
3) actual and relative density of shortleaf pine (*Pinus echinata* Mill.) has decreased, and
4) the ground cover has changed from predominantly grasses and forbs to leaf litter and some shade tolerant forbs.

In recent years, many federal, state, and non-governmental organizations began using prescribed fire as a tool to restore Missouri Ozark forest stands to historic conditions. In fact, during the spring of 1999, state and federal agencies in Missouri treated 69,000 acres of land with prescribed fire (George Hartman, personal communication). The principal objectives of these treatments were vegetation management, exotic species control, and enhancement of native species; little emphasis was given to fuel loadings or fuel reduction. At the same time, fuel loads are increasing across the Ozarks of southern Missouri, primarily due to fire suppression and the accumulation of slash from timber harvests. Concern is increasing regarding fuel accumulation in many areas. Fuel reduction treatments, including prescribed fire and thinning, can benefit forest management in the Ozarks by modifying stand structure to meet residual density objectives and reducing the chance or intensity of wildfire. Thus, it seems that the same tools, currently employed for very different objectives, may in fact accomplish multiple goals if properly applied across the landscape.

A quantitative evaluation of fuel reduction treatments in the Missouri Ozarks, to include change in forest structure, fuel structure and loading, cost effectiveness of treatments, and the ecological response to treatment, provides an opportunity to develop detailed fuel and fire behavior data for the oak-hickory forest type of the eastern deciduous forest. It also facilitates the evaluation and development of fuel reduction methods for the Missouri Ozarks, allows for parameterization and validation of fire behavior models, and allows for the quantification of fuel loads in a forest type that lacks fuel inventory data. Further, it allows land managers to predict the vegetative response to these treatments based on pre-treatment conditions and, therefore, tailor management to achieve the desired outcome. The treatments being examined are prescribed fire, overstory thinning, and their interaction. Because topographic position affects not only vegetation but also fire behavior, data were collected across three broad topographic classes: protected or north-facing slopes, exposed or south-facing slopes, and ridge tops. In order to keep site- to site-variation to a minimum, the three replicated study blocks were maintained within the same land type association in southern Missouri.

Commonly occurring species in the pretreatment understory are presented in tables 2 and 3. The lack of recent disturbance on these sites is evidenced by the widespread occurrence of shade tolerant, fire intolerant species such as red maple (*Acer rubrum* L.), dogwood (*Cornus florida* L.), and black gum (*Nyssa sylvatica* Marsh.) across most topographic positions and sites. Some species such as bare-stemmed tick trefoil (*Desmodium nudifolia* (L.) DC.) were ubiquitous, occurring in nearly every plot across all sites. Other common species such as white oak (*Quercus alba* L.) and black oak (*Q. velutina* Lam.) were more prevalent on the more drought prone ridges and exposed slopes. Based on the experience of local resource managers, an immediate and marked response is expected from the initial treatments scheduled to be applied during the 2002 growing season.

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Table 2.—*Five most common woody species by topographic position and treatment block*

SITE 1		SITE 2		SITE 3	
Protected		**Protected**		**Protected**	
Acer rubrum	Red maple	*Acer rubrum*	Red maple	*Acer rubrum*	Red maple
Nyssa sylvatica	Black gum	*Cornus florida*	Flowering dogwood	*Nyssa sylvatica*	Black gum
Cornus florida	Flowering dogwood	*Sassafras albidum*	Sassafras	*Carya texana*	Black hickory
Carya tomentosa	Mockernut hickory	*Quercus alba*	White oak	*Sassafras albidum*	Sassafras
Sassafras albidum	Sassafras	*Quercus coccinea*	Scarlet oak	*Corylus americana*	American hazelnut
Ridge top		**Ridge top**		**Ridge top**	
Sassafras albidum	Sassafras	*Quercus alba*	White oak	*Sassafras albidum*	Sassafras
Cornus florida	Flowering dogwood	*Quercus velutina*	Black oak	*Corylus americana*	American hazelnut
Acer rubrum	Red maple	*Carya texana*	Black hickory	*Carya texana*	Black hickory
Diospyros virginiana	Common persimmon	*Nyssa sylvatica*	Black gum	*Quercus alba*	White oak
Quercus alba	White oak	*Sassafras albidum*	Sassafras	*Rhamnus caroliniana*	Carolina buckthorn
Exposed		**Exposed**		**Exposed**	
Sassafras albidum	Sassafras	*Sassafras albidum*	Sassafras	*Cornus florida*	Flowering dogwood
Acer rubrum	Red maple	*Quercus alba*	White oak	*Sassafras albidum*	Sassafras
Nyssa sylvatica	Black gum	*Nyssa sylvatica*	Black gum	*Nyssa sylvatica*	Black gum
Quercus velutina	Black oak	*Acer rubrum*	Red maple	*Quercus velutina*	Black oak
Carya tomentosa	Mockernut hickory	*Quercus velutina*	Black oak	*Quercus alba*	White oak

*Top 5 regenerating woody species (<1.6" dbh) by site and topographic position, based on total percent coverage across plots.

Table 3.—*Five most common ground flora species by topographic position and treatment block.*

SITE 1		SITE 2		SITE 3	
Protected		**Protected**		**Protected**	
Desmodium nudiflorum	Bare-stemmed tick trefoil	*Desmodium nudiflorum*	Bare-stemmed tick trefoil	*Desmodium nudiflorum*	Bare-stemmed tick trefoil
Acer rubrum	Red maple	*Acer rubrum*	Red maple	*Acer rubrum*	Red maple
Parthenocissus quinquefolia	Virginia creeper	*Cornus florida*	Flowering dogwood	*Parthenocissus quinquefolia*	Virginia creeper
Nyssa sylvatica	Black gum	*Sassafras albidum*	Sassafras	*Nyssa sylvatica*	Black gum
Cornus florida	Flowering dogwood	*Quercus alba*	White oak	*Carya texana*	Black hickory
Ridge top		**Ridge top**		**Ridge top**	
Desmodium nudiflorum	Bare-stemmed tick trefoil	*Quercus alba*	White oak	*Parthenocissus quinquefolia*	Virginia creeper
Parthenocissus quinquefolia	Virginia creeper	*Quercus velutina*	Black oak	*Sassafras albidum*	Sassafras
Rhus radicans	Poison ivy	*Carya texana*	Black hickory	*Corylus americana*	American hazelnut
Sassafras albidum	Sassafras	*Nyssa sylvatica*	Black gum	*Carya texana*	Black hickory
Cornus florida	Flowering dogwood	*Desmodium nudiflorum*	Bare-stemmed tick trefoil	*Desmodium nudiflorum*	Bare-stemmed tick trefoil
Exposed		**Exposed**		**Exposed**	
Sassafras albidum	Sassafras	*Vaccinium vacillans*	Lowbush blueberry	*Cornus florida*	Flowering dogwood
Desmodium nudiflorum	Bare-stemmed tick trefoil	*Sassafras albidum*	Sassafras	*Desmodium nudiflorum*	Bare-stemmed tick trefoil
Acer rubrum	Red maple	*Quercus alba*	White oak	*Sassafras albidum*	Sassafras
Parthenocissus quinquefolia	Virginia creeper	*Nyssa sylvatica*	Black gum	*Vaccinium vacillans*	Lowbush blueberry
Nyssa sylvatica	Black gum	*Acer rubrum*	Red maple	*Nyssa sylvatica*	Black gum

*Top 5 ground flora species by site and topographic position, based on total relative cover of each species by plot.

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PATTERNS OF FOREST LAND OWNER PARTICIPATION IN THE WEST VIRGINIA FOREST STEWARDSHIP PROGRAM

B.M. Jennings, D.J. Magill, D.W. McGill, and J. Warren[1]

West Virginia is the Nation's third most forested State in terms of percent forest cover with at least 80 percent of its land forested. Despite this wealth of timber, in the late 19th and early 20th century the State was almost completely cut over with 30 billion board feet being harvested between 1870 and 1920. Currently, there are numerous second growth stands that are approaching 100 years in age. This maturing timber and strong hardwood stump-age prices have given good reason for the forest industry to procure timber and private woodland owners to be enthusiastic about the potential uses of West Virginia's forests.

Anecdotal evidence and preliminary inventory figures suggest that the rate of cutting in the last decade has increased substantially. This situation of strong prices and strong demand leave non-industrial private forest land owners with good negotiating options. However, in order to take advantage of this opportunity landowners must make wise management and marketing decisions. Under the West Virginia Forest Stewardship Program (FSP), landowners have the opportunity to obtain cost-share dollars to support long-term forest management planning on their forested properties. In this research note we describe patterns of participation in the West Virginia Forest Stewardship Program over its first decade.

We used standard graphical methods and summary statistics to describe the forest stewardship properties in the State of West Virginia. The program is slightly over 10 years old, but we have restricted our analyses to the first decade only.

The Forest Stewardship Program began its initial operation in the central portion of the State. It was established in four central counties on a trial basis in order to assess landowner interest. In the first year over 31 thousand acres enrolled in the program. Such high enrollment confirmed a positive reception by landowners in West Virginia. As a result of this successful trial period, the FSP was made available to the entire State.

During the first decade of the program, 558 thousand acres were enrolled throughout West Virginia. Following 1991 annual enrollment increased through the first 5 years and peaked in 1996 when a total of approximately 105 thousand acres were added to the program. A comparison of the six districts that make up the State show that the Eastern Panhandle District accounts for 32 percent of the enrolled acres (fig. 1). The Central District contains the second largest amount with 20 percent of the total acres, followed closely by the Southwest District with 19 percent of the total acres enrolled.

We are currently investigating the relationship between property taxes, urbanization, and FSP enrollment. The Eastern Panhandle District is often considered a "bedroom community" of the Baltimore/DC metropolis, hence it is important to examine the seemingly significant relationship between FSP acreage and urbanization in this district. The acreages enrolled in the FSP are owned mostly by West Virginia residents (81 percent), but 19 percent of plan acres are owned by out-of-state residents. Records indicate that there are over 100 thousand acres enrolled in the FSP by out-of-state property owners (table 1). Nearly 50 thousand acres in forest stewardship plans are located in the Eastern Panhandle region. Maryland and Virginia residents account for the majority of out-of-state owners enrolled in the program. Surprisingly, FSP participants with mailing addresses in Florida rank third in enrolled acres.

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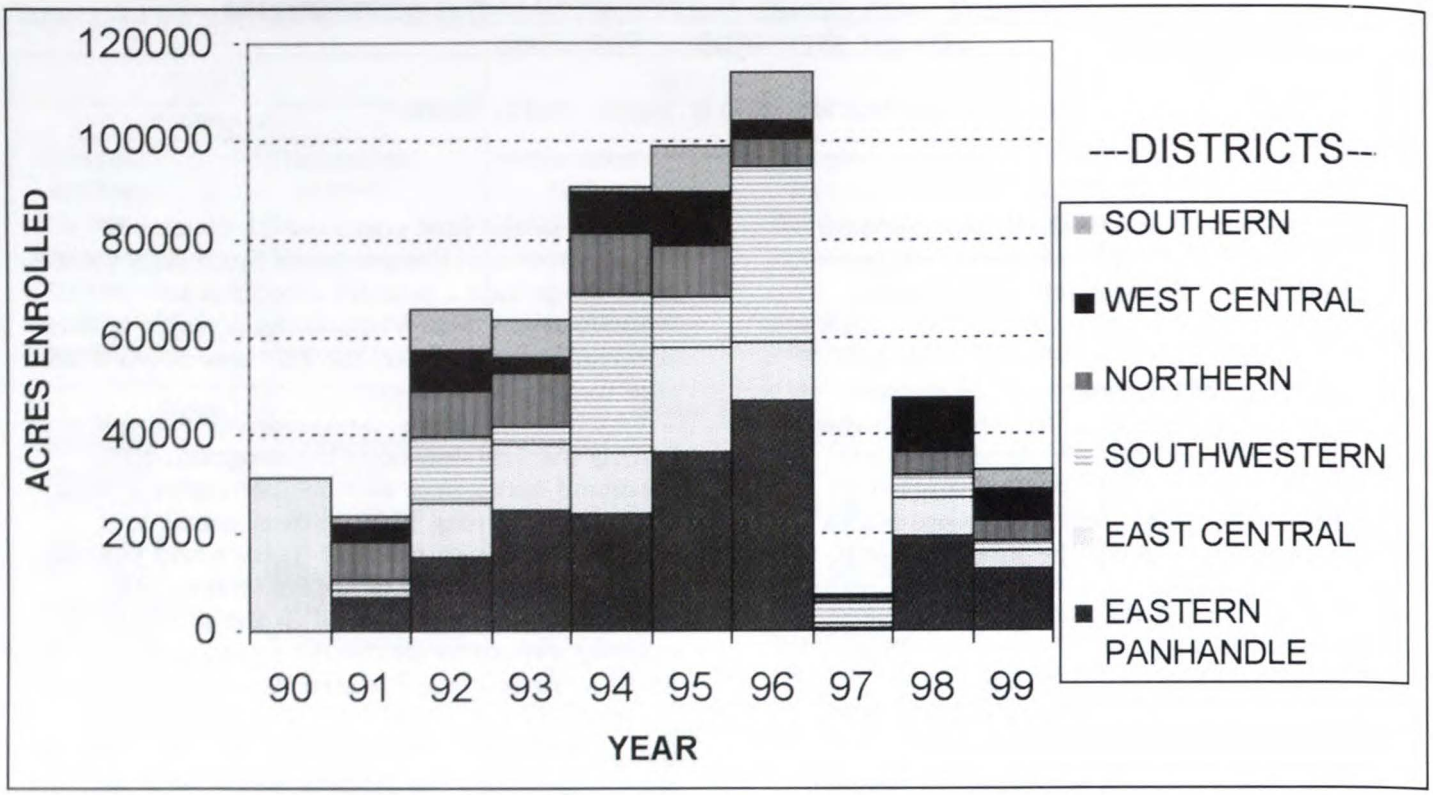

Figure 1.—*Ten-year pattern of the number of acres enrolled in the Forest Stewardship Program by District.*

Table 1.—*Total acreage of out-of-state landowners enrolled in the Forest Stewardship Program by district over the first decade. Acreages represent properties of program participants with mailing addresses outside of West Virginia. Districts correspond to West Virginia Division of Forestry administrative units. States are sorted by total acreage. States listed are those with residents with total ownership greater than 7,000 acres in the Forest Stewardship Program.*

	DISTRICTS						
State	**Northern**	**Eastern Panhandle**	**East Central**	**Southern**	**South Western**	**West Central**	**Total**
MD	4,232	24,328	3,342	916	54	3,076	35,947
VA	1,640	14,853	2,657	4,578	172	1,381	25,280
Other	2,565	4,243	4,456	3,157	1,797	5,740	21,958
FL	164	0	3,071	294	2,850	1,594	7,972
OH	450	1,850	1,302	621	970	2,482	7,675
PA	203	3,320	351	394	1,410	1,503	7,180
Total	9,253	48,594	15,178	9,960	7,253	15,775	106,014

A NEW BOOK: THE ECOLOGY AND SILVICULTURE OF OAKS

Paul S. Johnson, Stephen R. Shifley, and Robert Rogers[1]

A new book, *The Ecology and Silviculture of Oaks*, is now available (Johnson and others 2002). The book focuses on the oaks of the United States and was written for forest and wildlife managers, ecologists, silviculturists, environmentalists, students of those fields, and others interested in sustaining oak forests for their many tangible and intangible values. Although the approach of the book is fundamentally silvicultural, it is based on the premise that effective and environmentally sound management and protection of oak forests and associated landscapes should be grounded in ecological understanding.

Much has been written about the ecology and silviculture of oaks. This literature represents, in one sense, an informational "embarrassment of riches." This stems from a previous lack of synthesis within and across two broad fields of study: ecology and silviculture. Although the literature on North American oaks dates to the colonial period, most of it was written within the last 50 years. Much literature resides in relatively obscure scientific and technical journals, proceedings, government publications, and other sources that can be difficult to locate and retrieve. The first 12 proceedings of the Central Hardwood Forest Conferences are important sources for much relevant knowledge (Fralish 2002). But even with ready access to this information, a synthesis into a holistic framework has, until now, been unavailable.

One of the objectives of the book is to present ecological and silvicultural concepts that can be used to address an array of problems defined by various perceptions of how we should treat oak forests. The current trend in managing forests and forested landscapes is away from narrowly defined timber and other commodity objectives and toward a broader philosophy of sustaining desired ecological states. The book accordingly is intended not so much as a how-to-do-it management manual as it is a source of ideas on how to think about oak forests as responsive ecosystems. Armed with that understanding, we believe managers and conservators of oak forests will be better able to adapt to changing social values and to simultaneously build and act on co-evolving ecological and silvicultural knowledge.

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WATER, WATER EVERYWHERE... INTEGRATED RIPARIAN RESEARCH IN THE NORTH CENTRAL REGION

Lynne M. Westphal[1]

ABSTRACT.—Riparian areas are important for many reasons including timber, residential areas, biodiversity, recreation, and industry. Because of the intricate land/water interface, these lands require special care. Determining what this care should be requires more knowledge than we currently have. Hence, the North Central Research Station (NCRS) has initiated the Sustaining Riparian Landscapes Integrated Program. NCRS scientists and their partners investigate the impacts of different riparian delineation and quantification methods and the impacts of human actions from development to restoration. Research projects include regeneration of oak bottomlands, landscape impacts of best management practices, landowner understanding of riparian health, and restoration of Rust Belt landscapes.

In the North Central Region of the United States, we have the Great Lakes and great rivers. We have thousands of smaller lakes and smaller rivers, rural riparian areas and urban ones. People are drawn to riparian areas for many reasons: we play in them, we obtain resources from them to build our homes and communities, we seek solace and retreat in them, we value the wildlife and plants that riparian areas support, we rely on their role in maintaining water quality. Because of these critical needs and the resulting policy and management challenges riparian areas present, the North Central Research Station (NCRS) has embarked upon the Sustaining Riparian Landscapes Integrated Research and Development Program.

The North Central Research Station has a long history of riparian research, including bottomland hardwood research and the work of the river recreation research unit in the 1970s. These examples are based in a disciplinary approach to research, in these cases silviculture and recreation research. But rarely are policy questions and conundrums rooted in a single discipline, rather these issues are often multi- and inter-disciplinary. To make headway that is meaningful for science, policy, and management, we need to tie together information from the social, biological, and physical sciences. At the North Central Research Station we are addressing this need to tie it all together through the Sustaining Riparian Landscapes Integrated Program (Riparian Program). In addition, the Station is using the integrated program approach to address landscape change and forest productivity issues.

WHY A RIPARIAN PROGRAM?

Few regions of the country have a greater predominance of riparian areas, or more potential for conflict among their uses, than the North Central Region of the United States. Water, and the surrounding resources, are particularly important in this region. For example, 37 percent of forest land in northern Minnesota is within 57 m of surface water. More broadly, the seven national forests of the upper Great Lakes states make up only 4 percent of total Forest Service lands, yet they contain 41 percent of Forest Service lakes, reservoirs, and ponds. Lakes, open-water wetlands, rivers, and streams are the settings for intense residential, recreational, and industrial development. The importance of riparian areas in the region is clear: our economy, our sense of place, our quality of life depends on riparian areas.

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Riparian area issues vary significantly across the North Central Region. Forests in the southern tier of states in the North Central Region are largely Central Hardwood forests, and the riparian systems tend to be rivers and streams rather than lakes. While abundance of riparian areas characterizes the northern states, loss of riparian forest from human impacts may be the dominant issue in the Central Hardwood riparian areas. An estimate based on USDA Forest Service Forest Inventory data is that 4 percent of the Central Hardwood forests in Missouri, Iowa, Illinois, Indiana, and Ohio are in riparian areas. There used to be more. Conversion to agriculture has been a major force in the dwindling amount of riparian forests in the Central Hardwoods. But in some areas, we are seeing a reversal of this trend as former agricultural bottomlands are returning to forest either by default or through active management. These few bottomland forests are among the most diverse of the Central Hardwood forests in terms of number of species. This species richness is just one of the important characteristics of riparian forests.

Central Hardwoods are important in urban centers, too. For example, oak savanna is an important ecosystem in the Chicago metropolitan area and a primary focus of the Chicago Wilderness coalition. This umbrella organization seeks to restore and promote the native biodiversity of the lands at the southwest corner of Lake Michigan, from northwest Indiana to southeast Wisconsin. Oak was a dominant forest type here, and, like in bottomlands and other oak forests, regeneration is a key issue.

This is why the North Central Research Station has embarked on the Riparian Program: water resources in the North Central Region are of great importance. Our goal is to better understand riparian ecological services, as well as the broader array of societal benefits gained from these areas. Ultimately, our aim is to better inform management and policymaking decisions on riparian land use from the northern coniferous forests to the Rust Belt to the Central Hardwood forest.

CENTRAL RESEARCH QUESTION

The central question guiding the Riparian Program is:
How does diverse land use in the North Central Region change riparian area structure, impact ecological functions, and influence the benefits people derive from riparian resources?

This question gets at the heart of the riparian issue in the North Central Region. North Central riparian areas not only have great ecological value, they also are sources of substantial economic and social benefits from recreation, residential development, timber management, agriculture, wildlife, and industry. These land uses usually threaten ecological value, but in some cases may actually help to restore function (e.g., rehabilitation of agricultural land through forestry, or the development of greener industry).

In our attempt to answer the central question, we identified these three critical problem areas to guide our scientists and research partners:

1) Quantifying riparian areas, their extent, function, and benefits
2) Impacts to riparian areas, investigating factors affecting function and value
3) Rehabilitating riparian areas

These problem areas are described below, with examples of current research addressing each issue.

PROBLEM AREA 1: QUANTIFYING RIPARIAN AREAS, THEIR EXTENT, FUNCTION, AND BENEFITS

A fundamental constraint to understanding riparian land use issues in the North Central Region is the lack of good assessments of the resource at appropriate scales. We need to better understand the amount and ecological condition of riparian areas associated with different types of aquatic systems and in different landscape settings from the Central Hardwoods to the northwoods, and how these variables have changed over time. To get to this understanding, we first must develop sound methodologies to delineate riparian areas at different scales (e.g., watershed or stand) based on ecological properties. We can't know the amount and condition of riparian areas until we understand just which lands *are* riparian, and we don't have precise estimates of how much forest is riparian in the Central Hardwoods or other types of forest in the North Central Region.

At the same time that we expand and refine the delineation methods, we must also consider the economic and social benefits people gain from different riparian land uses, the tradeoffs among riparian benefits individuals and communities are willing to accept, and the social and economic consequences of riparian land use conversion. Projects in this problem area involve quantification of riparian areas from

different perspectives, including ecologically based delineations; indexing of lakeshore development potential; monitoring of recreation use; and assessment of stakeholder perceptions of riparian health. Two projects are outlined below: the North Central Region Riparian Assessment and a study of the contribution of ancient wood to riparian health.

How Much Land is Riparian? The North Central Riparian Assessment

Brian Palik, USDA Forest Service, NCRS, Grand Rapids, MN and Dave Heinzen, Minnesota Department of Natural Resources, Grand Rapids, MN, lead this research. They are trying to answer the question of how do we determine what is and is not riparian? The typical approach has been a fixed width buffer, say 30 or 60 m from the water's edge. But does this approach, based more on available data and expediency rather than ecological realities, best delineate riparian areas? Do policies for riparian areas get applied to the right lands? Both over- and under-application of policy and management techniques for riparian areas could have significant ecological, economic, and social implications. Different methods of delineating riparian areas can give wildly different figures. For instance, Illinois land can be less than 2 percent or over 10 percent riparian, depending on the delineation method used. The riparian assessment seeks to clarify these issues and to develop meaningful definitions and delineation tools for riparian areas.

Taking the first step in the assessment, Brian Palik worked with partners to compare a variety of methods for delineating riparian areas, including fixed widths (30, 60 m), soils (indicators of hydric soils), geomorphology (slope and elevation), and the extent of floods. The intent was to explore approaches having greater links to functional ecotones using existing data layers. This was done at two scales of analysis—the entire North Central Region and a finer grained analysis of two watersheds in Minnesota.

The 30 and 60 m buffers on streams, lakes, wetlands; hydric soil delineations; 100 year flood extent where available; and landuse/landcover analysis have been completed for the entire region. Now, Brian Palik and Dave Heinzen are developing more detailed geomorphic delineation approaches on two Minnesota watersheds. These data are (or very soon will be) available for use on the North Central Web site (http://www.ncrs.fs.fed.us/).

Ancient Wood Uncovered

Daniel Dey, USDA Forest Service, NCRS, Columbia, MO; and Richard Guyette, University of Missouri, Columbia, MO lead this research. They are studying the dynamics of coarse woody debris in streams in the North Central Region to improve our understanding of how forests and aquatic systems interact and to improve our management of riparian forests. They discovered that many streams in northern Missouri contain a substantial amount of very old wood in their channels. As the streams meander across their flood plains, they uncover ancient wood—often entire trees with their root systems—that have been preserved in 30 feet or more of alluvial soil. By using radiocarbon (^{14}C) and tree-ring dating methods, they have found in-stream coarse woody debris that ranged in age from less than 100 years before present (B.P.) to 11,900 years B.P. Furthermore, this ancient wood is not rare—wood older than 500 years B.P. was common in all the streams surveyed. The presence and abundance of ancient wood in Missouri streams was previously unknown.

Discovering wood this old is a dendrochronologist's dream come true, but ancient wood is also important today for stream and forest biodiversity. It provides habitat for aquatic invertebrates, contributes nutrients to the stream, and sequesters carbon. In fact, these ancient trees might have been one of the largest carbon sinks of their day. This raises a new ecological impact of straightening rivers and streams—have we undermined an important carbon sink? Would restoration of stream meanders and riparian forests be a useful component in a larger global change strategy?

This research provides new perspectives on the long-term dynamics of aquatic coarse woody debris and the complexity of how riparian forests and streams interact. Such information can be used to assess changes in riparian forest productivity, carbon sequestration, and forest composition as they relate to changes in climate and human use of these areas since the last ice age.

PROBLEM AREA 2: IMPACTS TO RIPARIAN AREAS, INVESTIGATING FACTORS AFFECTING FUNCTION AND VALUE

When we use things, we usually change them. This is certainly true in riparian areas. These changes (building boat houses, converting forest to agricultural production, straightening rivers, using water for cooling manufacturing equipment)

impact the riparian structure and function, and can impact the sustainability of the very uses and benefits we expect from riparian areas. We need to understand the impacts of these changes across different land uses and in different landscape settings. With this understanding, we can then investigate means to mitigate impacts on riparian and aquatic resources and develop ways to balance multiple benefits within riparian areas. Several current and new Station-led efforts focus on riparian alteration through research into riparian forest management approaches, efficacy of riparian buffers, characteristics of riparian forest fragments, and watershed-scale land use impacts. Two studies addressing these issues are outlined below: a study looking at landscape level impacts of actions at the stand level and an investigation of lake homeowners' perceptions of riparian health.

Assessment of Timber and Wildlife Resources in Missouri Riparian Management Zones as Affected by Best Management Practices (BMPs)

Hong S. He, Department of Fisheries and Wildlife Science, University of Missouri, Columbia, MO; and Steve Shifley, USDA Forest Service, NCRS, Columbia, MO, lead this research. How can we understand potential effects of various forest management techniques at the landscape level? Modeling can help. One current project will analyze the effects of different methods of designating riparian management zones (e.g., as fixed-width buffers vs. buffers based on streamside topography and hydrology) on the size of riparian zones and their contributions to timber and wildlife. We will investigate how the riparian designation method and the application of Missouri's riparian forest BMPs affect the pattern and structure of the larger forest landscape (including riparian and other forests). Within the riparian zones we will assess how the application of BMPs affects forest age structure and timber production over time.

Specifically, this research will look at three or more representative landscapes (from 3,000 to 10,000 acre watersheds) along a major river system in Missouri. In each landscape, we will designate alternative riparian management zone configurations including fixed width riparian zones and functional riparian zones based on topographic features. We will simulate the outcome of applying BMPs within riparian zones during timber harvesting and examine predicted future vegetation age structure within riparian zones and across the entire watershed. The results will be compared with the alternative of not applying BMPs within riparian zones during timber harvesting. Results will also be analyzed within the context of the entire watershed where simulation of upland management alternatives has already been completed.

Landowners' Understanding of Impacts in Riparian Areas May Lack Scientific Basis

Pamela Jakes, USDA Forest Service, NCRS, St. Paul, MN, leads this research. When northern Wisconsin landowners were asked about the impacts of their actions on the health of riparian areas, their perceptions were very different from what science tells us. Landowners were not aware of many of the basic ecological principles that are critical to sustainable land use and how their decisions regarding home and road construction and landscaping, or their choice of recreational activities may impact the environment. The findings suggest that scientists and their colleagues involved in public outreach need to develop ways to better communicate findings regarding potential human impacts on riparian ecosystems and "healthy" alternative land use practices. Managers also need to work with those involved in developing land use policy, such as county zoning or Best Management Practices, so that guidelines and regulations reflect the best science available. This communication is especially critical in riparian areas that are experiencing increased development.

PROBLEM AREA 3: REHABILITATING RIPARIAN AREAS

Restoration or rehabilitation of ecological function and social value is a primary concern in degraded riparian areas. Research needs vary from development and testing of technologies, methods, and materials, to assessing the social implications of rehabilitation and restoration. Severe degradation, seen in many agricultural and urban landscapes of the Central Hardwood Region, often calls for intensive efforts to rehabilitate functions. Functional restoration may not necessarily translate into restoration of original ecosystem characteristics (e.g., vegetation cover or large wetland systems). Several current North Central Research Station research projects focus on riparian restoration or rehabilitation, including development and testing of plant materials, stream-crossing restoration, reforestation of agricultural bottomlands, and balancing of ecological and economic needs in the revitalization of urban riparian areas. Two studies are outlined below: one addressing damaged riparian areas in a Rust Belt landscape, the other looking at the regrowth of oaks in the Missouri bottomlands.

Revitalizing the Rust Belt: The Calumet Initiative

Sandy Verry, USDA Forest Service, NCRS, Grand Rapids, MN; and Lynne Westphal, USDA Forest Service, NCRS, Evanston, IL; lead this research. The Calumet Region of northwest Indiana and northeast Illinois is a quintessential Rust Belt landscape, one of many communities struggling in the former industrial belt that ran from Pittsburgh to Chicago. Many industries still thrive in Calumet and, paradoxically, threatened and endangered species also flourish there in remnant natural areas of oak and wetland nestled among used and abandoned industrial plants. The Calumet Region also draws recreationists hoping to see the rare bird, catch the big fish, or just enjoy the outdoors. The North Central Research Station, along with its many and diverse partners, is working to help local and regional planners and managers decide how to move the area toward ecological and economic health. Our role is to provide information and technology and to facilitate in these efforts.

One research project that epitomizes this intersection of ecological and economic health is the Indian Creek project. Indian Creek is a creek in name only: it is a ditch and provides no habitat for fish or other aquatic life, and its bottom is deep in muck from slag. It was formed as people filled in over 300 acres of the Hyde Lake wetlands. About 40 acres of wetland remain, with the "creek" running adjacent to it. The filled lands have been used for heavy industry and manufacturing waste disposal for decades; currently they are abandoned brownfields. Sites like this can't be restored to pre-settlement conditions—it is financially impossible to remove acres of slag and there is nowhere to put it that would not cause the same environmental injustice somewhere else. But it is possible to increase the ecological health of many of these sites, including Indian Creek and the remnant Hyde Lake wetland.

Sandy Verry has studied the geomorphology of this "creek," creating 36 cross sections to develop a longitudinal profile of the creek including estimates of its bankfull flow, water sources, water velocity, and other critical stream characteristics. With this information, Sandy developed plans for re-engineering Indian Creek's shape and depth to provide pools, riffles, and other aquatic structure. Dan Dey is planning studies to help revegetate Indian Creek's banks—now slag and ash and not hospitable for growing things.

Since this project began, the Ford Motor Company has purchased land neighboring Indian Creek and will develop the first North American Supplier Park Campus on these brownfields. Sandy Verry and other North Central Research Station scientists have contributed to the design plans for this industrial site and Ford has committed to rehabilitate the portion of Indian Creek that runs through its property. Ford plans to take advantage of the creek as an aesthetic aspect of the Supplier Park design, with offices, day care centers, and other components of the Supplier Park using the creek (e.g., office views of the creek, picnic tables adjacent to the creek). North Central Research Station scientists continue to be involved in the development of both the Supplier Park and Indian Creek and will monitor the ecological impacts of the rehabilitation project, leading the way to new techniques for re-creating ecological integrity in places humans have abused.

Regenerating Oak Bottomlands

Daniel Dey, USDA Forest Service, NCRS, Columbia, MO, leads this research. He is examining oak regeneration, an ongoing concern, on productive bottomlands in Missouri, where croplands are reverting back to forests. Oak forests are important for many reasons, including wildlife habitat, native biodiversity, recreation, forest products, and healthy ecosystem processes and functions. In one ongoing project, a new nursery product (RPM™ seedlings), soil mounding, and a cover crop of redtop grass are being assessed for their ability to improve oak seedling survival. The RPM™ seedlings have significantly larger root systems than typical bare root seedlings. The stock planted in this study had 17 to 20 g of dried root in the standard bare root stock compared to 120 to 140 g in the RPM™ stock. Because of the increased root structure, significantly more oaks from the RPM™ stock survived, with some trees even producing acorns in the second year after transplant. Soil mounding and redtop grass were less significant in increasing survival. When complete, this study will produce guidelines to help regenerate native hardwood forests in Missouri and other Central Hardwood states.

WATER, WATER EVERYWHERE

The riparian areas in the North Central Region face many demands. We turn to these lands at the water's edge for everything from timber production to serenity. Riparian areas in the Central Hardwood forests face challenges similar to all of North Central's riparian areas:

definitions and delineation methods need to be refined and made workable on the ground, the wide ranging importance of these riparian areas to people needs to be better understood and placed in a management and policy context, and impacts of actions at the stand or housing development level need to be understood in the wider landscape context. But the Central Hardwood riparian areas also face some unique challenges. So much forest land at the water's edge has been converted to other uses, primarily agriculture and urban development, that we particularly need to understand the lost function and develop tools for restoration and rehabilitation of these riparian areas.

Because of our high expectations of riparian landscapes, and because of the impacts of our past actions, riparian areas in North Central require a sustained initiative developing information that can be readily used by policymakers and managers as they make critical decisions for these important landscapes. The North Central Research Station is providing this research initiative in its Sustaining Riparian Landscapes Integrated Research and Development program. We are investigating the impact of diverse land use in the region on riparian area structure, ecological functions, and the benefits people expect from riparian resources. In this way, we hope to be a part of science for people's sake, and to make a difference for the ecological health of our region and the quality of life for the people who live, work, and play here.

ACKNOWLEDGMENTS

This paper draws on the charter of the Sustaining Riparian Landscapes Integrated Research and Development Program, written by Brian Palik and Nancy Walters, and on project summaries written by the project principal investigators mentioned in each project description.

WATERSHED SCALE ASSESSMENT OF THE IMPACT OF FORESTED RIPARIAN ZONES ON STREAM WATER QUALITY

J.A. Webber, K.W.J. Williard, M.R. Whiles, M.L. Stone, J.J. Zaczek, and D.K. Davie[1]

ABSTRACT.—Federal and state land management agencies have been promoting forest and grass riparian zones to combat non-point source nutrient and sediment pollution of our nations' waters. The majority of research examining the effectiveness of riparian buffers at reducing nutrient and sediment inputs to streams has been conducted at the field scale. This study took a broader watershed approach in assessing the stream water quality benefits of forested riparian buffer zones. Stream nitrate, orthophosphate, and ammonium concentrations were compared in 11 agriculturally impacted watersheds with varying percentages of riparian forest area. The study catchments are located in the Sugar Creek watershed in southern Illinois and contain similar soil types and land uses (predominately row crop agriculture). The Geographic Information System software, ArcView 3.2 (ESRI), was used to delineate the Percent Forested Area in the Riparian Buffer Zone (PFARBZ), within 15 m of the stream across the total length of the 1st and 2nd order streams within each catchment. Stream water samples were collected at baseflow over a 9-month period for nutrient analysis. Preliminary results suggest that watersheds with greater PFARBZ have lower dissolved stream PO_4^{-3}-P concentrations. However, no significant relationship was found between PFARBZ and dissolved stream NH_4^+-N and NO_3-N concentrations.

Throughout the Midwestern United States, agriculture has been and continues to be the predominant land use. Agriculture is considered a major contributor to stream degradation in the United States, having adverse effects on in-stream habitats and water quality (USEPA 1994). Nutrient loading and sediment deposition have been identified as the water quality problems of greatest concern (Chester and Schierow 1985).

Over the past few decades, studies have been conducted to examine the amount of nutrients carried in runoff, factors that affect nutrient runoff, and how nutrient export can be minimized (Lowrance and others 1983, Peterjohn and Correll 1984, Pinay and Decamps 1988, Muscutt and others 1993, Edwards 1996). One tool that has become widely accepted to reduce excess nutrient and sediment export is vegetated riparian buffer zones.

The primary determinants of the volume and kinetics of external inputs into a riparian zone are derived from the watershed's land use, size, and gradient. Recent literature suggests that riparian buffer zones can facilitate water quality improvements in different ways depending on how water passes through the buffer zone (Muscutt and others 1993, Correll 1996, Herron and Hairsine 1998). A forested riparian zone can effectively filter lateral overland flow when it enters the buffer strip as sheet flow draining from a small field with a minimal slope of 5 percent or less (Peterjohn and Correll 1984). In addition, several studies have shown that riparian

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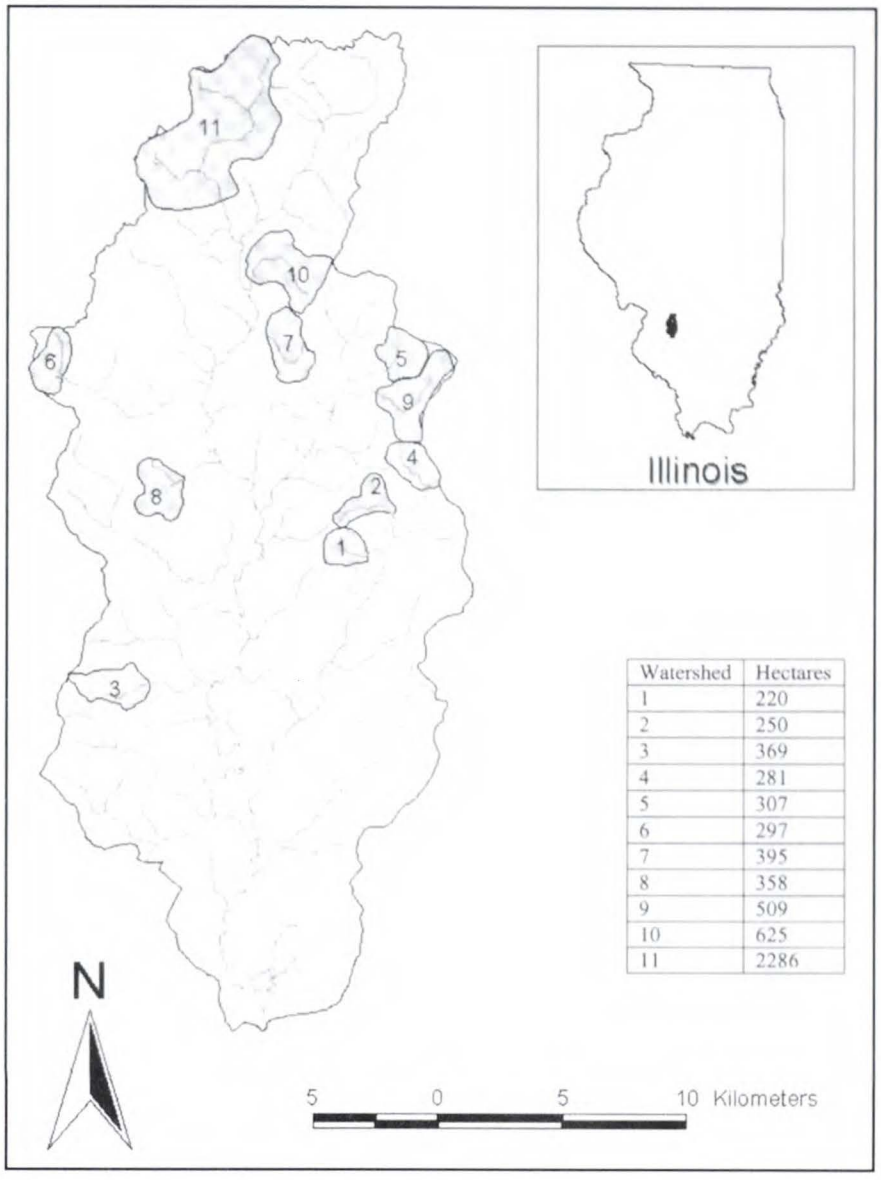
zones can decrease the nutrient levels of groundwater entering streams (Jacobs and Gilliam 1985, Cooper 1990, Haycock and Pinay 1993, Jordan and others 1993).

The majority of research aimed at quantifying the water quality benefits of forested riparian buffer zones has been conducted at the field scale (Groffman and others 1991, Haycock and Pinay 1993, Addy and others 1999). Herron and Hairsine (1998) suggested that land management initiatives need to be directed at the catchment scale if riparian buffers of realistic widths are to be effective. This study examined the effectiveness of forested riparian zones at attenuating agricultural nutrient inputs to streams at the watershed scale. We hypothesized that nutrient concentrations would decrease as percent forested area within riparian buffer zones (PFARBZ) increased.

METHODS

Eleven headwater watersheds with varying percentages of riparian forest cover were located within the Illinois Department of Natural Resources (IDNR) Sugar Creek Pilot Watershed, Clinton, Bond, and Madison Counties, Illinois, in the fall of 2000 (fig. 1). The 11 watersheds were selected using 1998 Digital Ortho-photo Quadrangles (DOQ's; 1 m resolution). We found the predominant land-use on all of the watersheds to be row crop agriculture, and assumed



Watershed	Hectares
1	220
2	250
3	369
4	281
5	307
6	297
7	395
8	358
9	509
10	625
11	2286

Figure 1.—*Location of study watersheds within the Sugar Creek drainage, Illinois.*

that nitrogen and phosphorus fertilizer application rates were similar across the study catchements.

County stream image data files provided by the Illinois Natural Resource Geospatial Data Clearinghouse (INRGDC) were used to identify the stream channels (INRGDC 2000). All visible drainage networks, 1st and 2nd order streams, within the watershed were also digitized and added to the total stream length. A 15-m buffer zone was created on each side of a stream, and the total area within the 15-m buffer zones was derived. The presence of forested area within the 15-m riparian zone was also digitized from the DOQ's using ArcView 3.2 (ESRI). The total area of forest within the 15-m buffer zone was divided by the total area of the 15-m buffer zone to estimate PFARBZ. Riparian forest species composition was primarily early successional species.

All but one of the study watersheds was less than 700 ha in area (fig. 1). The watersheds contained a variety of soil types and complexes within the riparian areas including: Wakeland silt loam, Blackoar silt loam, Blair silty clay loam, Darmstadt-oconee complex, Cowden-Piasa complex, Virden silt loam, Muren silt loam, Hickory clay loam, Oconee silt loam, Wakeland silt loam, Fayet silt loam, Virden-Piasa silty loam, and Rozetta silty clay loam.

Stream water grab samples were collected monthly (May-December 2001) during baseflow conditions. Due to the intermittent nature of some of the study streams in September and October, 81 of the possible 88 samples (11 streams * 8 sampling dates) were taken. The samples were collected at a medium depth in the middle of the stream cross-section. The stream sampling point was located at the defined mouth of each stream (fig. 1). Samples were transported on ice to the Forestry Laboratory of Watershed research (FLOW) at Southern Illinois University, Carbondale, for analysis.

Stream samples were analyzed for pH, conductivity, and dissolved orthophosphate (PO_4^{-3}-P), nitrate (NO_3^--N), and ammonium (NH_4^+-N) concentrations. Dissolved orthophosphate (standard method 8178) and dissolved ammonium (standard method 8048) were measured using a Hach 4000D spectrophotometer. Dissolved nitrate was analyzed on a Dionex Ion Chromatograph Series 4000i. Conductivity and pH were measured using a Fisher Scientific AR 20 pH/conductivity meter. Results were compiled and averaged by site for the eight collection dates. Regression analysis was performed with each of the water quality parameter means and PFARBZ using the JMP in 4 Academic software package from the SAS Institute, Cary, NC. Significant relationships were tested at the α=0.05 level.

RESULTS

The ranges of PO_4^{-3}-P, NH_4^+-N, and NO_3^--N concentrations (mg L^{-1}) across the 11 watersheds were 0.000 to 11.709, 0.010 to 12.860, and 0.042 to 4.440, respectively. PFARBZ exhibited a significant (p=0.009) negative relationship with, and explained 50 percent of the variation in dissolved PO_4^{-3}-P concentrations across the 11 study watersheds (fig. 2). No significant relationships were found between PFARBZ and dissolved NH_4^+-N or NO_3^--N concentrations across the study watersheds (figs. 3 and 4). In addition, stream pH and conductivity were not significantly related to PFARBZ.

DISCUSSION

Numerous investigations have reported that naturally established riparian buffers reduce the impact of agriculturally derived nonpoint source pollutants in surface waters (Lowrance and others 1984, Peterjohn and Correll 1984, Jacobs and Gilliam 1985, Jordan and others 1993, Synder and others 1998). Several studies have shown that riparian forests can decrease phosphorus concentrations in overland flow from adjacent row crop fields (Dillaha and others 1989, Gilliam 1994, Daniels and Gilliam 1996, Lyons and others 1998). Results from this study suggest that percent forested buffer area at the watershed scale is an adequate predictor of dissolved phosphate concentrations. These findings support those of Jones and others (2001), who found that the percentage of forested stream miles within a watershed explained 63 percent of the variation in dissolved stream phosphorus levels in the Mid-Atlantic region of the United States.

Forested riparian zones have also been shown to reduce dissolved nitrate concentrations in groundwater entering the riparian zone from adjacent row crop fields (Lowrance and others 1984, Peterjohn and Correll 1984, Jacobs and Gilliam 1985, Cooper 1990, Schipper and others 1991, Osborne and Kovacic 1993, Synder and others 1998). However, our study did not show a significant relationship between PFARBZ and dissolved stream nitrate concentrations (fig. 4). James and others (1990) also found that

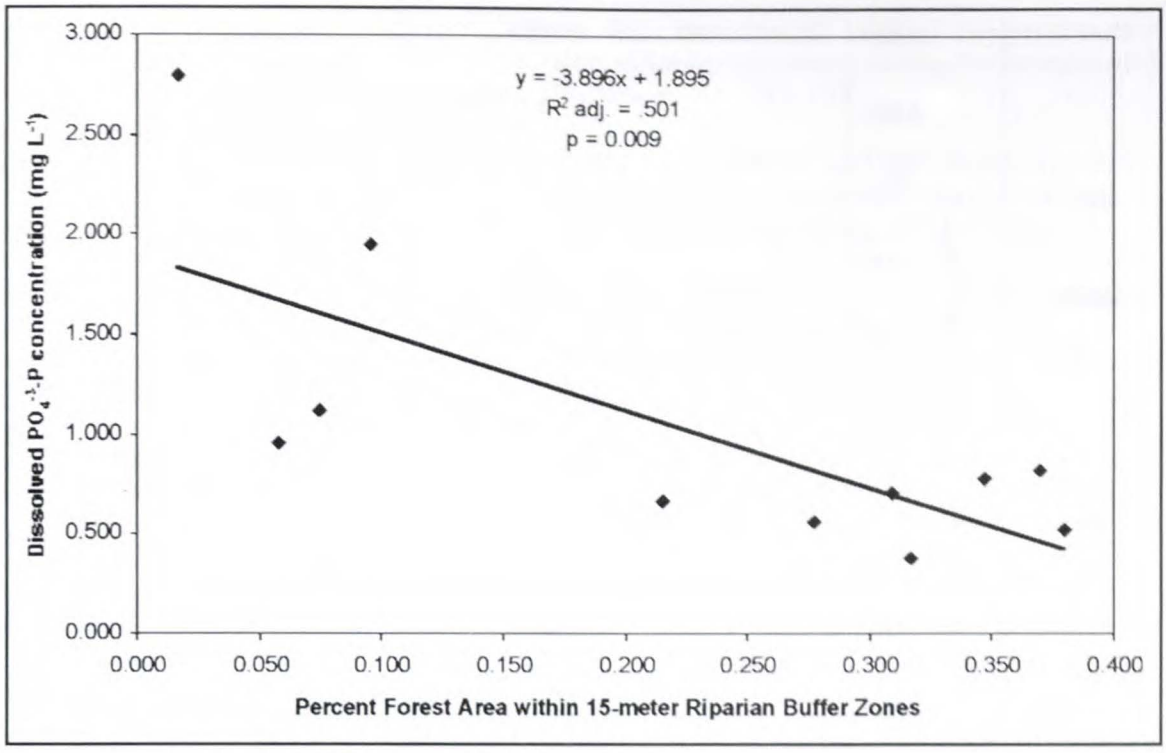


Figure 2.—*Dissolved PO_4^{-3}-P concentration versus percent forest area within 15-m riparian buffer zones.*

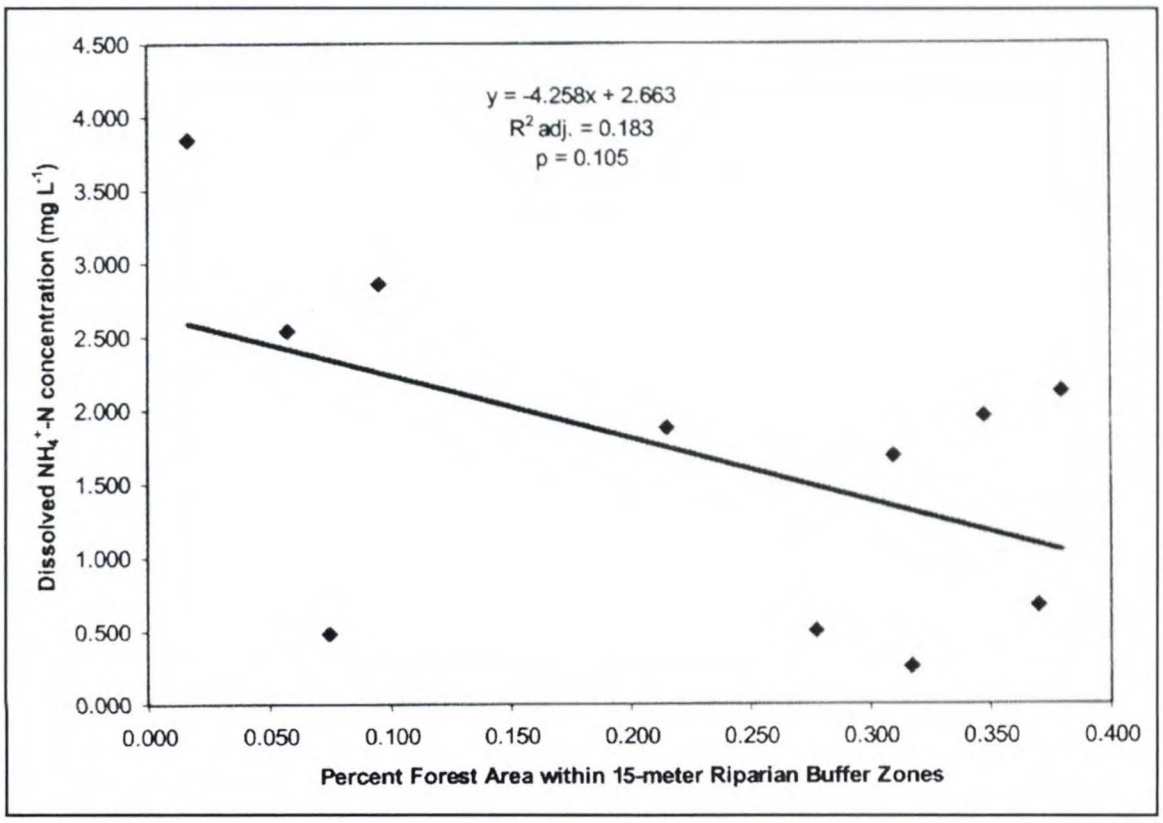


Figure 3.—*Dissolved NH_4^+-N concentration versus percent forest area within 15-m riparian buffer zones.*

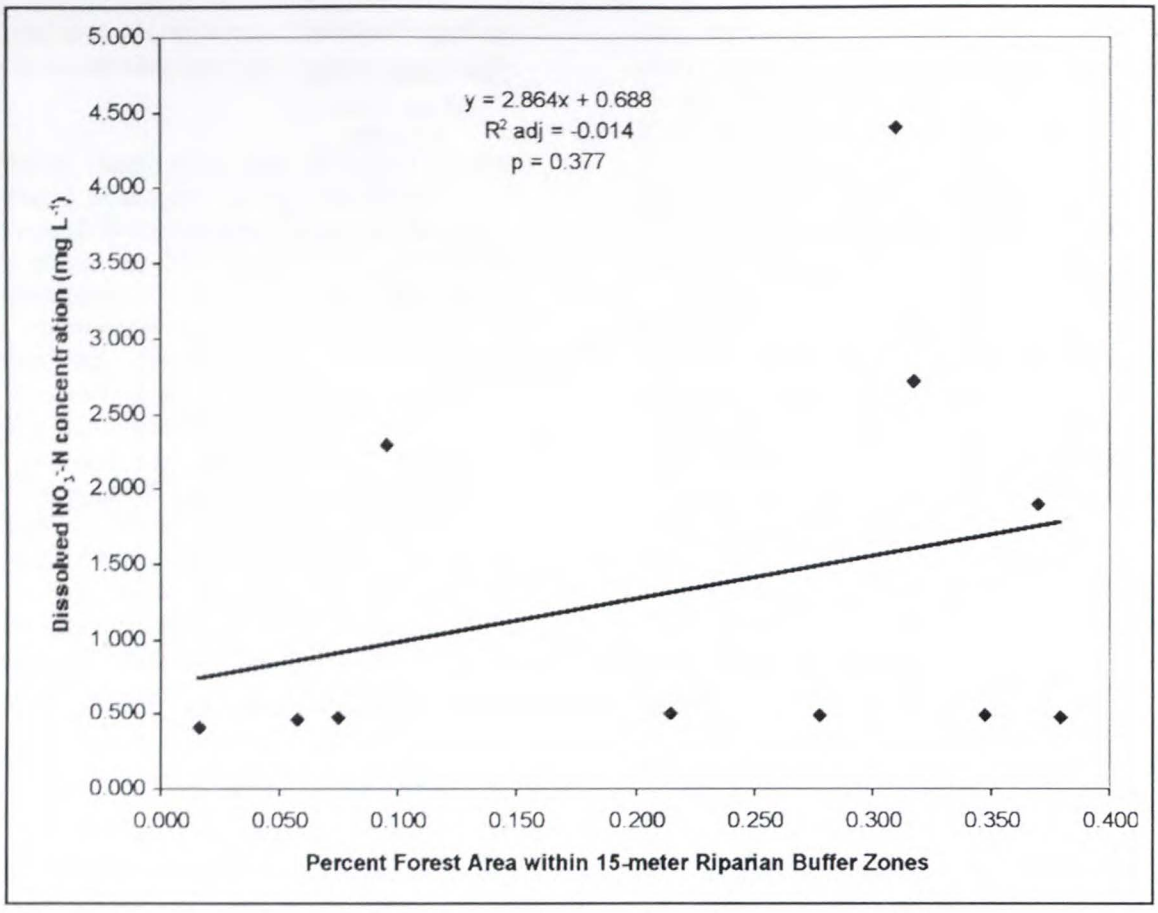


Figure 4.—*Dissolved NO_3-N concentration versus percent forest area within 15-m riparian buffer zones.*

riparian forests did not reduce dissolved nitrate concentrations in groundwater.

Prior studies have shown that forested riparian zones can reduce ammonium concentrations entering as surface flow from adjacent agricultural lands (Peterjohn and Correll 1984, Daniels and Gillaim 1996). Our results show that dissolved ammonium concentrations were not significantly correlated with PFARBZ (fig. 3), but showed a non-significant trend that would become significant by eliminating one point, a site with low PFARBZ and low ammonium concentration. Aside from this site, ammonium concentrations decreased with increasing PFARBZ.

A potential reason for the lack of stronger relationships between PFARBZ and stream dissolved nitrogen species (NO_3^--N and NH_4^+-N) is that some or all of our study watersheds could be nitrogen saturated. The theory of nitrogen saturation was developed in the forestry literature to explain the existence of increased dissolved nitrogen export from forested watersheds within the past 2 decades (Agren and Bosatta 1988, Aber and others 1989, Van Miegroet and others 1992). Aber and others (1989, 1998) described nitrogen saturation as a series of stages where the capacity of an ecosystem's soil, microbes, and vegetation to retain nitrogen is gradually depleted due to chronic nitrogen inputs (atmospheric deposition). This may be occurring on our agricultural study watersheds due to nitrogen fertilizer applications, which are substantially higher than atmospheric deposition levels.

The observed non-relationship between PFARBZ and dissolved stream nitrogen concentrations could also be due to significant variability in riparian soil and in-stream denitrification rates among the 11 study watersheds. Several studies have shown riparian soil and in-stream denitrification rates to be highly variable within a watershed and among adjacent watersheds (Hill and Sanmugadas 1985, Cook and White 1987, Hill 1988).

CONCLUSIONS

Our study indicates that PFARBZ can be used to assess the effectiveness of forested riparian buffer zones at attenuating some nutrients at the watershed scale. PFARBZ exhibited a significant negative relationship with dissolved PO_4^{-3}-P concentrations, but did not show a significant relationship with dissolved NH_4^+-N or NO_3^--N concentrations, although there was a trend of lower ammonium in streams with higher PFARBZ. This lack of a relationship with dissolved stream nitrogen concentrations could be an indication that riparian buffer zones are nitrogen saturated or it may be due to differences in soil and in-stream denitrification rates in the study watersheds. Increasing the number of 1st and 2nd order streams sampled may prove to strengthen presented relationships. Future watershed scale analyses of the effectiveness of riparian zones could be expanded to include watersheds with riparian zones of differing species composition and ages.

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ASSESSING THE EXTENT OF NITROGEN SATURATION IN NORTHERN WEST VIRGINIA FORESTED WATERSHEDS: A SURVEY OF STREAM NITRATE CONCENTRATIONS

Karl W.J. Williard, David R. DeWalle, and Pamela J. Edwards[1]

ABSTRACT.—Twenty-seven forested watersheds in northern West Virginia were sampled for stream nitrate concentrations during summer 1997 and fall 1998 baseflow periods to determine if Fernow watershed 4, an often-cited and studied nitrogen saturated basin, was anomalous or regionally representative in terms of stream nitrate levels. Baseflow stream NO_3-N concentrations ranged from 0.005 to 1.54 mg L^{-1}, with nearly half of the streams being higher than Fernow 4 in both summer and fall. Variations in stream nitrate were related to basin geology, whereas no direct effects of basin physiography could be established. It was concluded that Fernow 4 was representative of regional stream nitrate concentrations and the majority of the watersheds surveyed showed similar signs of nitrogen saturation.

Northern temperate forests have long been considered to be nitrogen limited ecosystems, yielding low concentrations of nitrogen in streamflow. This theory has been challenged by researchers who observed increasing levels of nitrogen export from forested headwater streams (Aber and others 1989, Driscoll and others 1989, Johnson and Lindberg 1992, Murdoch and Stoddard 1992, Stoddard 1994). Researchers have coined the phrase "nitrogen saturation" to describe a forest ecosystem's declining ability to retain nitrogen (Aber and others 1989, Agren and Bosatta 1988). In nitrogen saturated systems, inorganic nitrogen is supplied in excess of microbial and vegetative demand (Aber and others 1989, 1998; Stoddard 1994).

Stoddard (1994) described nitrogen saturation in a series of stages based on seasonal stream nitrate concentrations. In stage 0, stream nitrate-N concentrations are near zero year round. In stage 1, nitrate-N becomes elevated (0.5 to 0.8 mg NO_3-N L^{-1}) in the dormant season due to a lack of microbial and vegetative uptake. Stage 2 is characterized by elevated stream nitrate-N concentrations year round. Watershed 4 at the Fernow Experimental Forest has been cited as the best example of a stage 2 nitrogen-saturated watershed in the northeastern United States, exhibiting elevated year round nitrate-N concentrations (0.72 ± 0.10 mg NO_3-N L^{-1} from 1997 to 1998) (Peterjohn and others 1996, Fenn and others 1998, Williard 1999).

The Fernow Experimental Forest is located in northern West Virginia, an area that receives high inputs of bulk atmospheric nitrate deposition (5.0 kg NO_3-N ha^{-1} yr^{-1} from 1983 to 2001) (P. Edwards – written comm.). Watershed 4 at Fernow is a control watershed that has been exporting high rates of nitrate (5.1 kg NO_3-N ha^{-1} yr^{-1}) for the past two decades (Adams and others 1993). On Fernow 4, nitrate-N bulk deposition inputs (5.0 kg ha^{-1} yr^{-1}) and stream outputs (5.1 kg ha^{-1} yr^{-1}) are equal and outputs show little seasonal variation. Both of these characteristics are indicators of nitrogen saturation (Gilliam and others 1996, Peterjohn and others 1996).

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Fernow also contains two other control catchments in close proximity (within 1.1 km) to Fernow 4: watersheds 10 and 13. Nitrate-N exports from Watershed 13 (4.0 kg NO_3-N ha^{-1} yr^{-1}) and Watershed 10 (1.2 kg NO_3-N ha^{-1} yr^{-1}) are lower than Watershed 4, even though they receive similar amounts of atmospheric nitrogen deposition (P. Edwards, written communication). All three of these watersheds contain deciduous vegetation with forest stand ages between 60 and 90 years old. If atmospheric deposition were leading to nitrogen saturation, these watersheds would be expected to export similar amounts of nitrate. This raises the question of whether other factors such as geology/soil chemistry, physiographic parameters, and past land use history are causing differences in stream nitrate export in this localized area.

Our objective was to examine the spatial and temporal variability of stream nitrate concentrations in an area that contains a nitrogen saturated watershed, Fernow 4. The specific null hypotheses tested were:

1) growing and dormant season stream nitrate concentrations in 26 surrounding forested watersheds were similar to Fernow 4, and
2) dormant season stream nitrate concentrations were greater than growing season nitrate concentrations in 27 forested watersheds.

Assessing the local variability in stream nitrate concentrations in relation to geology and watershed characteristics may aid in the identification of local controls on stream nitrate concentrations in forested watersheds.

STUDY SITES

Study watersheds were located on the east and west side of Shavers Fork River in close proximity (within 10 km) to the Fernow Experimental Forest to keep climatic conditions and nitrogen deposition rates as uniform as possible (fig. 1). The 27 watersheds were all located between the elevations of 537 m and 1,136 m, which was the range of elevation on one of the largest watersheds (Canoe Run). Gilliam and Adams (1996) found that wet ammonium and nitrate deposition rates were not significantly different at three sampling sites (689 to 838 m elevation) within and near the Fernow Experimental Forest.

The watersheds selected for the study were all 100 percent forested. Predominant overstory species found on the study watersheds were red oak (*Quercus rubra* L.), sugar maple (*Acer saccharum* Marsh.), red maple (*Acer rubrum* L.), yellow-poplar (*Liriodendron tulipifera* L.), and American beech (*Fagus grandifolia* Ehrh.). West and south facing slopes contained white oak (*Quercus alba* L.) instead of red oak. Eighteen of the twenty-seven study watersheds contained actively managed stands, which have experienced timber harvesting, but none have been completely clearcut. The most recent partial cutting on any of the study watersheds was 4 years prior to sampling.

METHODS

Field Measurements

Stream water grab samples were taken on June 17, 1997, from 9:00 a.m. to 6:00 p.m. and on October 23, 1998, from 9:45 a.m. to 5:00 p.m. near the mouth of the watersheds during baseflow conditions. The samples were immediately put on ice for transport to the Water Analysis Laboratory at The Pennsylvania State University where they were analyzed within 48 hours for dissolved nitrate and ammonium using cadmium reduction and automated phenate methods, respectively (American Public Health Association 1995). Stream ammonium concentrations were near or below the detection limit (0.005 mg L^{-1}) in the study watersheds, which was expected given the lack of ammonium leaching through the soil matrix. Stream pH and specific electrical conductance (SEC) also were taken at each watershed using calibrated, portable meters (pHTestr 3 and TDSTestr, both by Oakton Inc., Singapore).

The 27 sampled watersheds were delineated on topographic maps (Parsons, Bowden, Elkins, and Montrose, WV 7.5 minute USGS quadrangles) and their bedrock geology types were determined with the use of quadrangle geology maps (Reger 1923, 1931). Represented bedrock types were grouped into two categories [(1) Catskill/Chemung/Pocono shale and sandstone, and (2) Mauch Chunk shale/Greenbrier limestone] based on relative differences in published aquifer pH values (Ponce and others 1979; Geyer and Wilshusen 1982; Taylor and others 1982, 1983). The catchment area, relief ratio, and average basin slope were calculated using the delineated watershed boundaries. Watershed area was determined using a digital planimeter (Sokkia Corp., Overland Park, KS). Relief ratio was calculated as the maximum elevation difference divided by the basin length measured along the long axis of the basin. Average basin slope was calculated using the Wentworth (1930) line-intersection method.

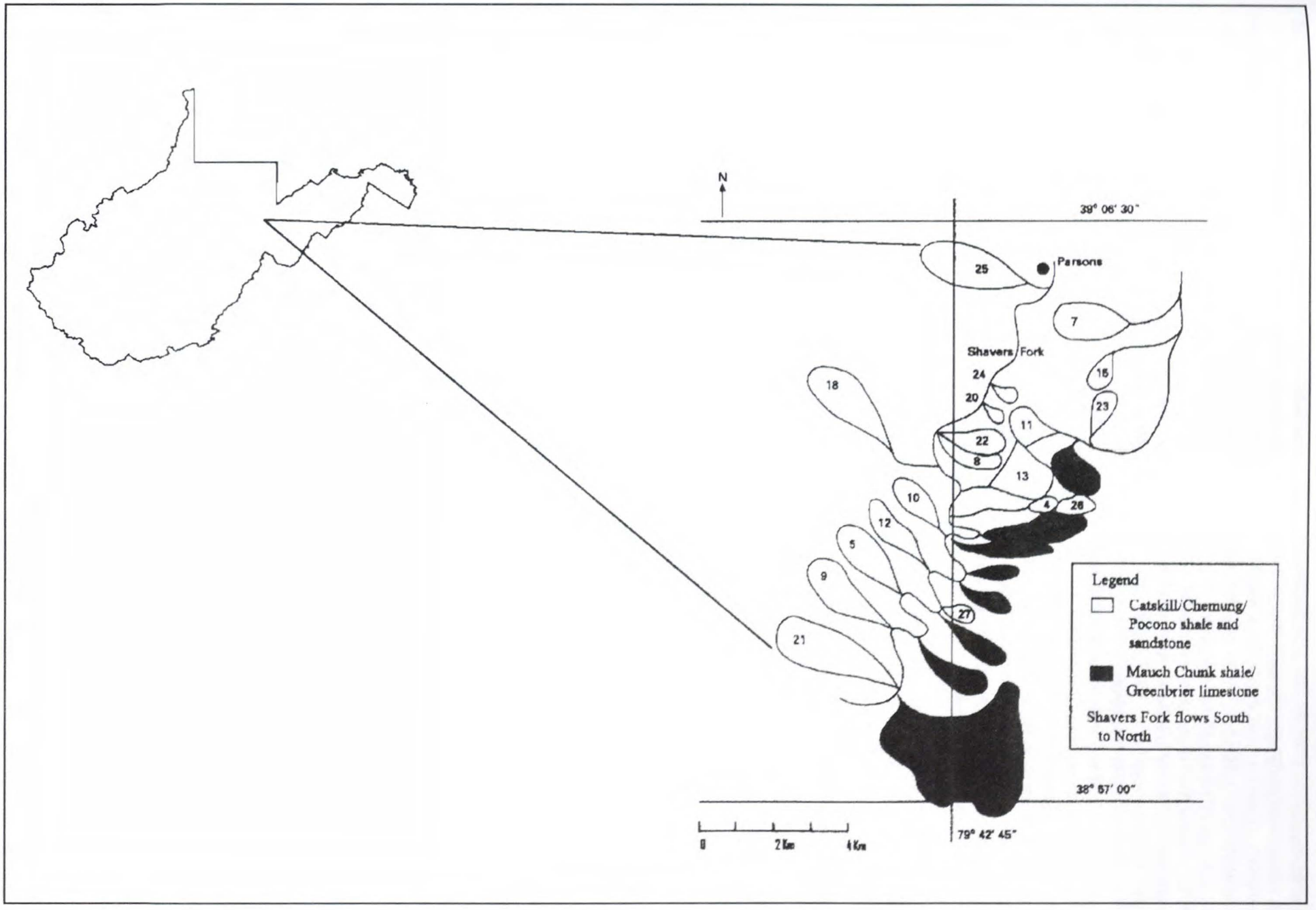

Figure 1.—*Locations of the 27 study watersheds in northern West Virginia.*

Statistical Methods

The SAS statistical package, Version 7 for Windows (SAS Institute Inc., Cary, NC), was used to analyze the data sets. Prior to performing any analysis of variance procedures (ANOVA) or t-tests, the data sets were analyzed to determine if the error terms were normally distributed and homogeneous (assumptions underlying ANOVA). Normal probability plots and Shapiro-Wilk test statistics confirmed that all the data sets were normally distributed. Bartlett's tests showed error variances for the various ANOVA models to be relatively homogeneous. Given the normal distribution of data and the homogeneity of error variance, no data transformations were required.

T-tests, assuming equal variances, were used to determine if mean summer and fall nitrate concentrations were significantly different at α=0.05. One-way ANOVA models were developed to test if summer and fall mean stream nitrate-N concentrations, SEC, and pH were significantly different between the two geology categories [(1) Catskill/Chemung/Pocono shale and sandstone and (2) Mauch Chunk shale/Greenbrier limestone). The two geology categories were assigned quantitative dummy variables (CCP=1, MCG=2) in linear regression models of summer and fall stream nitrate versus geology category. Pearson correlation coefficients were determined for measured summer and fall stream chemistry variables (NO_3, SEC, pH) and the watershed physiographic parameters (average basin slope, relief ratio, and area). Significant correlation coefficients were noted for p values < 0.05.

RESULTS AND DISCUSSION

Stream Nitrate-N Concentrations

Stream nitrate concentrations were measured in the summer and fall on 27 forested watersheds in close proximity to Fernow 4, a watershed that exports relatively high amounts of nitrate (5.1 kg NO_3-N ha^{-1} yr^{-1}) (Adams and others 1993). The 27 watersheds exhibited a wide distribution of NO_3-N concentrations (0.005 to 1.54 mg L^{-1}) at the local scale of the study (fig. 2). The range of NO_3-N concentrations approaches the range of NO_3-N export (0.04 to 5.15 kg $ha^{-1}yr^{-1}$) DeWalle and Pionke (1996) observed at a regional scale in their review of forested watersheds in the Chesapeake Bay region.

Our measured stream nitrate-N concentrations on Fernow 4 (0.63 and 0.56 mg L^{-1} NO_3-N) were close to the mean (0.72 ± 0.10 mg L^{-1} NO_3-N) and within the range (0.50 to 1.00 mg L^{-1} NO_3-N) of nitrate-N concentrations measured at the U.S. Forest Service gauging station during 1997 and 1998 (P. Edwards written communication). Fernow 4's stream nitrate-N concentrations were approximately in the middle of the distribution of nitrate concentrations among the 27 watersheds (fig. 2). There were 10 watersheds in both the summer and fall sampling periods that had greater stream nitrate concentrations than Fernow 4. Given that Fernow 4 has been cited as the best example of a nitrogen saturated watershed in the northeastern United States (Stoddard 1994, Peterjohn and others 1996, Fenn and others 1998), the study results indicate that nitrogen saturation could be relatively widespread among forested watersheds in the local study area.

Mean dormant (late October) season nitrate-N concentrations (0.59 ± 0.16 mg L^{-1}) were not significantly different from growing season (mid-June) nitrate concentrations (0.56 ± 0.09 mg L^{-1}). The late fall sampling was assumed to represent dormant season concentrations because the sampling occurred after leaf fall and stream temperatures (5.2 to 10.8°C) were already declining toward dormant season levels. Many of the watersheds, including Fernow 4, had elevated stream nitrate concentrations during both the dormant season and growing season (fig. 2). The elevated (> 0.5 mg NO_3-N L^{-1}) growing season nitrate concentrations indicate that these watersheds are at stage 2 of nitrogen saturation, where the capacity of the microbes and vegetation to retain nitrogen during the growing season has been exceeded (Aber and others 1989, Stoddard 1994). In non-saturated systems, nitrate leaching rates are expected to be lower in the growing season compared to the dormant season because of greater nitrogen demands by microbes and vegetation in the growing season.

Approximately one-half (14 of 27) of the watersheds had greater stream nitrate-N concentrations in the summer compared to the fall (fig. 2). This was not expected, given the significant vegetative uptake and microbial immobilization of nitrogen during the growing season. Perhaps there was significant N fixation rates occurring on some of these watersheds during the growing season, which resulted in greater nitrate leaching. Watersheds with greater summer stream concentrations were not of a particular geology type or geographically clustered in one area of the study region. Of the 18 study watersheds containing actively managed timber stands, 10

of them had higher stream nitrate concentrations in the summer. Thus, the presence of timber management does not help explain why certain watersheds have higher stream nitrate concentrations in the summer.

Repeated sampling of a large number of watersheds allows the assessment of whether a one-time baseflow survey is an adequate representation of relative stream nitrate concentrations. Specifically, this can be assessed by ranking summer and fall stream nitrate concentrations and examining if stream nitrate ranks change over time. Using a non-parametric sign test with an S table (Noether 1991), the median change in rank was found to be 3 with a 95 percent confidence interval of 1 to 5 (fig. 3). Thus, the 27 streams exhibited relatively little change in rank, which indicates that one-time baseflow sampling can serve as an adequate index to stream nitrate concentrations.

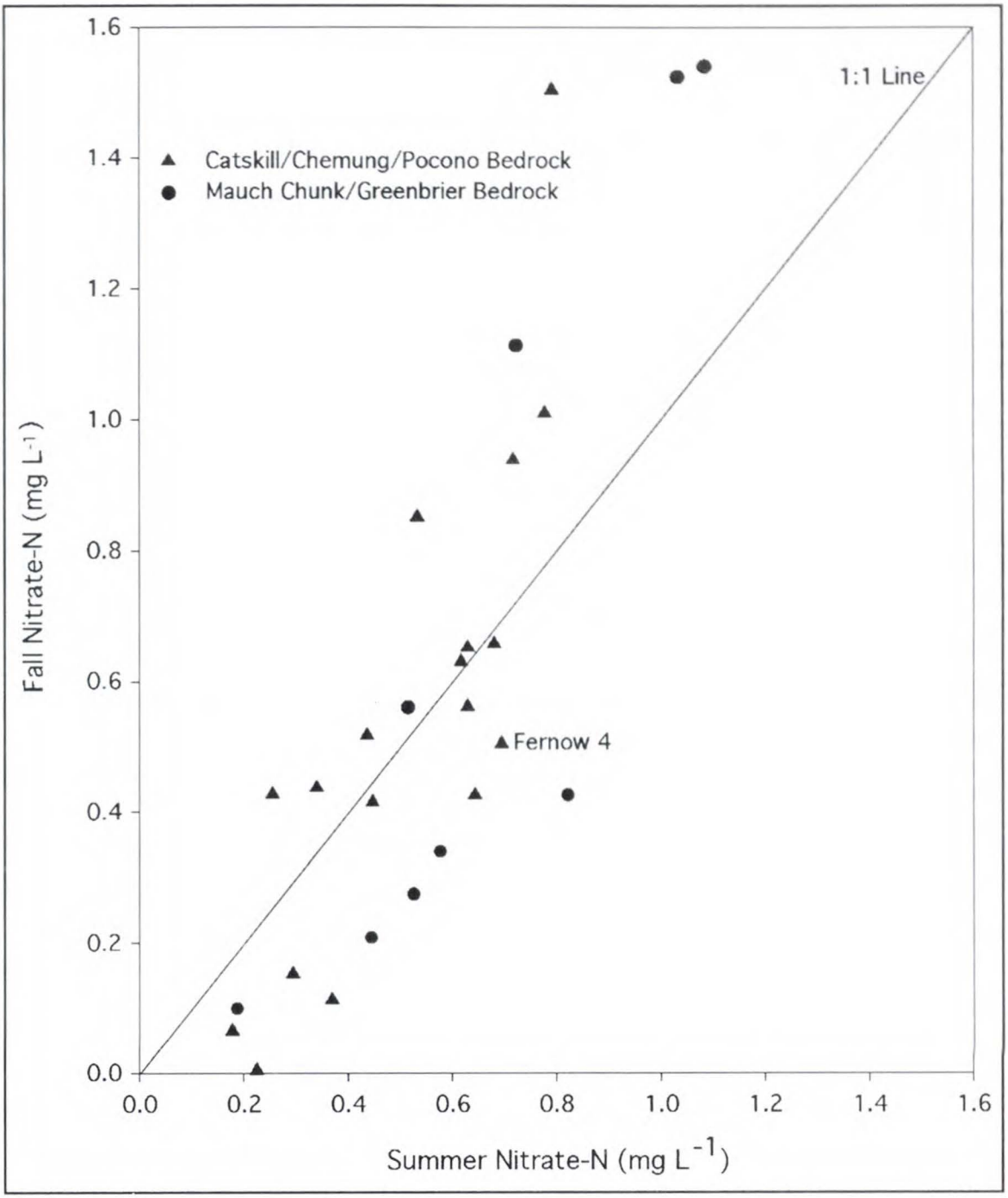

Figure 2.—*Fall stream nitrate-N concentrations for the 27 study watersheds.*

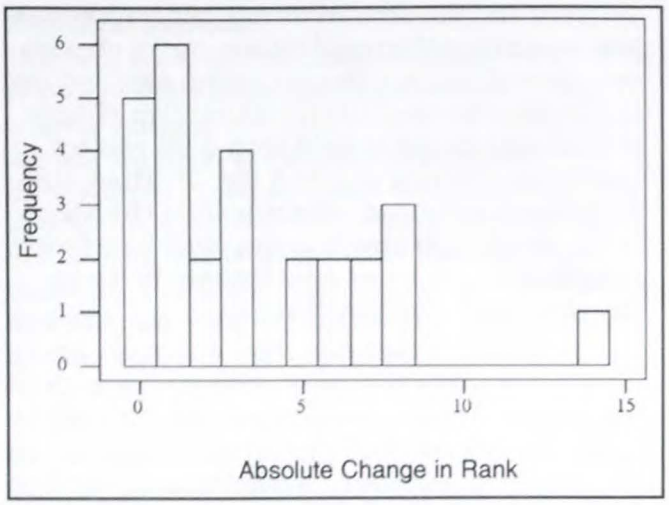

Figure 3.—*Change in stream nitrate-N ranks between summer and fall among the 27 forested watersheds.*

Explaining the Variability in Stream Nitrate-N Concentrations

Geology

Forested watersheds containing predominantly Mauch Chunk shale/Greenbrier limestone (MCG) bedrock (n=8) were found to have significantly greater mean summer stream nitrate concentrations than watersheds containing predominantly Catskill/Chemung/Pocono shale and sandstone (CCP) bedrock (table 1). Geology dummy variables (CCP=1, MCG=2) explained 18 percent of the variation in summer stream nitrate-N concentrations.

The observed stream nitrate differences among watersheds with different bedrock types agree with Ponce and others (1979) findings on tributaries of the Little Black Fork Creek, WV, which was one of the 27 study watersheds. Ponce and others (1979) reported that stream base cations, pH, and SEC differed among tributaries with different bedrock types. Ponce and others (1979) measured stream nitrate-N, but they did not analyze it as a variable that could differ by bedrock type. Nevertheless, Ponce and others (1979) raw data showed that stream nitrate-N did vary among three geology types contained in our study (Greenbrier, Mauch Chunk, and Catskill). Stream nitrate-N concentrations were relatively high on tributaries draining Greenbrier (1.31 mg L^{-1}) and Mauch Chunk (1.02 mg L^{-1}) bedrock types. The Catskill bedrock type yielded waters with significantly lower mean nitrate-N concentrations (0.69 mg L^{-1}), which agrees with our findings.

Even though there were mean differences between the two geology categories (MCG and CCP), there was significant variability in stream nitrate-N concentrations within each category. Fernow 4, 10, and 13 all contained 100 percent Catskill shale and sandstone bedrock yet Fernow 4 and 13 had significantly higher stream nitrate-N concentrations than Fernow 10. Thus, geology did not help explain stream nitrate concentration differences among the three Fernow watersheds.

The mechanism by which geology affects stream nitrate can only be speculated upon. Geology probably has an indirect control on nitrate leaching through its relationship with soil chemistry. Low soil pH and calcium have been noted to limit soil microbial activity and growth rates, respectively (Alexander 1977, Norris and others 1991, Smith 1995, Paul and Clark 1996).

The two represented geology categories likely have significantly different soil Ca and pH, based on the stream SEC and pH measurements and earlier findings by Auchmoody (1972). Mean summer and fall stream SEC levels were significantly higher among watersheds containing MCG bedrock (shale/limestone) compared to CCP bedrock (shale/sandstone) (table 1). Stream SEC generally is correlated well with Ca leaching rates, which are normally higher in watersheds with greater soil Ca pools. Further evidence for a geology/soil calcium link in our study is provided by Auchmoody (1972), who sampled forest soils from different geology types within Randolph and Tucker Counties, WV (our study location).

Table 1.—*Mean stream chemistry values for the two major geology categories*

Basin geology	NO_3-N (mg L^{-1})		pH		SEC (µS cm^{-1})	
	Summer	Fall	Summer	Fall	Summer	Fall
Catskill/Chemung/Pocono	0.50 a[1]	0.52 a	6.86 a	6.98 a	24.7 a	24.2 a
Mauch Chunk/Greenbrier	0.72 b	0.75 a	7.72 b	7.59 b	75.0 b	70.0 b

[1]Parameter means with different letters in a column are significantly different at α=0.05.

Auchmoody (1972) found that forest soils derived from both Greenbrier and Mauch Chunk bedrock had significantly greater calcium concentrations than soils derived from Catskill and Chemung bedrock. Watersheds with MCG bedrock had significantly higher mean summer and fall stream pH than watersheds with CCP bedrock (table 1). The MCG bedrock watersheds likely have higher mean soil pH values than the CCP watersheds, since relative differences in stream pH are generally correlated with differences in soil pH. Differences in soil pH and Ca levels may be causing the observed differences in stream nitrate concentrations among watersheds with MCG and CCP bedrock.

Physiographic Parameters

Summer and fall stream nitrate concentrations were not significantly ($\alpha = 0.05$) correlated with the physiographic parameters average basin slope, watershed relief ratio, and watershed area. Average basin slope did exhibit a weak correlation with summer ($r = 0.341$, $p = 0.081$) and fall ($r = 0.329$, $p = 0.093$) stream nitrate-N concentrations. Nitrate concentrations could be greater on steeper sloped watersheds because recharge water has shorter residence times. Thus, there is less opportunity for microbes and vegetation to assimilate the nitrate in the infiltrating water. Steeper sloped watersheds generally have more narrow riparian zones, which results in a lower percentage of anoxic soils. In these watersheds there would be less opportunity for denitrification, yielding higher nitrate concentrations. Average basin slope means were not significantly different between the two geology categories (MCG and CCP), indicating that geology effects did not confound the correlation of average basin slope with stream nitrate concentrations.

Past Land Disturbances

Nitrate leaching to streams can be affected by past land disturbances (logging, fire, farming) (Fenn and others 1998). Eighteen of the twenty-seven study watersheds were at least partially logged within the last 30 years, based on records from the Monongahela National Forest, WV (L. White, personal communication) and the Fernow Experimental Forest, WV (P. Edwards, personal communication). Four of the watersheds were partially cut as recently as 1993, 4 years prior to stream sampling. Clearcutting experiments on forested watersheds showed short term increases in stream nitrate concentrations (1 to 4 years), with nitrate export returning to pre-cutting levels by the third or fourth year due to nitrogen uptake by regrowing vegetation (Patric 1980, Hornbeck and others 1987, Lynch and Corbett 1991, Dahlgren and Driscoll 1994, Pardo and others 1995). Based on these findings, even the most recent partial cutting (4 years prior to sampling) on the study watersheds should not have impacted stream nitrate concentrations.

Previous fires also can affect nitrate leaching to streams by impacting soil nitrogen pools. Severe fire can volatilize nitrogen in the organic layer and upper mineral horizons, depleting long-term soil nitrogen pools (Raison 1979). There was no record of severe fire on any of the study watersheds. However, the effects of fire cannot be ruled out given the high frequency of slash-related fires after the initial logging of mid-Appalachian forests in the late 1800s to early 1900s.

Farming can either build-up or deplete soil N pools depending on the amount of fertilization that occurred. If soil nitrogen pools were severely depleted by past land disturbance, the pools still could be in an aggrading phase that would limit nitrate leaching to streams. The existence of past farming was not assessed on any of the study watersheds. However, it is likely that some of the watersheds were historically cultivated. In the early 1900s, many of the steep mid-Appalachian forested slopes were cleared and farmed until yields declined and they were left to revert to forest land (Lima and others 1977).

CONCLUSIONS

Fernow 4, a watershed cited to be nitrogen saturated, was representative of the local forested watersheds in terms of baseflow stream nitrate concentrations. Fernow 4 was just above the median stream nitrate concentration of the 27 watersheds sampled, indicating that nitrogen saturation may be a prevalent condition among local forested watersheds. The majority of the watersheds had elevated stream nitrate concentrations both in the fall and summer, symptoms of nitrogen saturation.

Differences in bedrock geology explained 18 percent of the variation in summer stream nitrate concentrations. Watersheds containing primarily Mauch Chunk shale/Greenbrier limestone had significantly greater summer stream nitrate concentrations than watersheds containing Catskill/Chemung/Pocono shale and sandstone. Geology likely had an indirect effect on stream nitrate concentrations, based upon its relationship with soil chemistry. Differences in soil

pH and Ca concentrations among the two geology categories could be causing the observed differences in stream nitrate leaching, given their documented effects on microbial nitrate production. Stream nitrate-N differences between Fernow watersheds 4, 10, and 13, which all contain Catskill bedrock, suggest that other factors such as past land use history (e.g., fire, farming, and logging) may also play a role in explaining stream nitrate variation at a local scale.

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RIPARIAN SHRUB BUFFERS REDUCE SURFACE WATER POLLUTANT LOADS

W.A. Geyer, C. Barden, K. Mankin, and D. Devlin[1]

Surface water resources in Kansas often contain concentrations of pesticides, nutrients, and sediments that are of concern to local citizens. The United States Geological Survey reported in 1999 that 97 percent of streams and 82 percent of lakes in Kansas would not fully support all uses as designated by state statutes (U.S. Geological Survey 1999). Bacteria and sediment contaminate a high percentage of streams while nutrients and pesticides contaminate many lakes.

One primary source of pollutants is water flow through watersheds from agricultural land (crop and grazing lands). The hydrology of streams and rivers in Kansas has been dramatically altered since settlement times in the 1800's. Cropland tillage practices, urban and transportation development, channel straightening, and livestock grazing practices have led to accelerated surface water movement, increased non-point pollution, and greater stream bank erosion.

Native riparian vegetative cover has been greatly reduced in much of the State. Reestablishing riparian buffers along streams could reduce flood damage and streambank erosion, improve wildlife habitat, and filter out pollutants including dissolved nutrients, pesticides, bacteria, and sediments.

The effectiveness of riparian buffers to remove pollutants from surface runoff is dependent on slowing sheet flow across the buffer, increasing infiltration, and prolonging subsurface flow. However, riparian buffers along agricultural fields in Kansas may not function effectively because physical barriers are inadequate to intercept the overland flow. In addition, surface flows are concentrated through terrace channelization or development of micro-relief near or in the riparian buffer. The later may result from sedimentation development at the entry of the buffer or flood deposits forming a levy on the bank of the stream.

There is little information available on the functionality of current riparian buffer areas in Kansas. Most of the riparian buffer research reported to date has been conducted in the Mid-Atlantic and southeastern United States (Lowrance and others 1985). The climate, soils, and hydrology of Kansas differ considerably from those on the eastern seaboard. Schultz and others (1994) have established several riparian studies in Iowa; however, important differences exist between Kansas and Iowa in the crop species grown, fertilizers and pesticides used, and in soil types and hydrology.

We have initiated a study to develop regional information on the effectiveness of riparian buffers to landowner and agency personnel in Kansas. This project will develop three riparian buffer demonstration and education sites across the State. Our objective is to evaluate the water quality-improving effectiveness of young, planted riparian buffers, and compare them to nearby native, mature, riparian woodland or grass-only filters. Monitoring equipment installed on the sites includes runoff samplers for both manual and automated collection of surface water samples and flow volume.

A shrub buffer demonstration and education site was established with American plum (*Prunus americana* Marsh.) in spring 2001. As will be done for all sites, we characterized the current vegetative cover and condition, soil properties, site characteristics, and past agricultural production practices. Surface runoff was sampled as water left the adjacent crop

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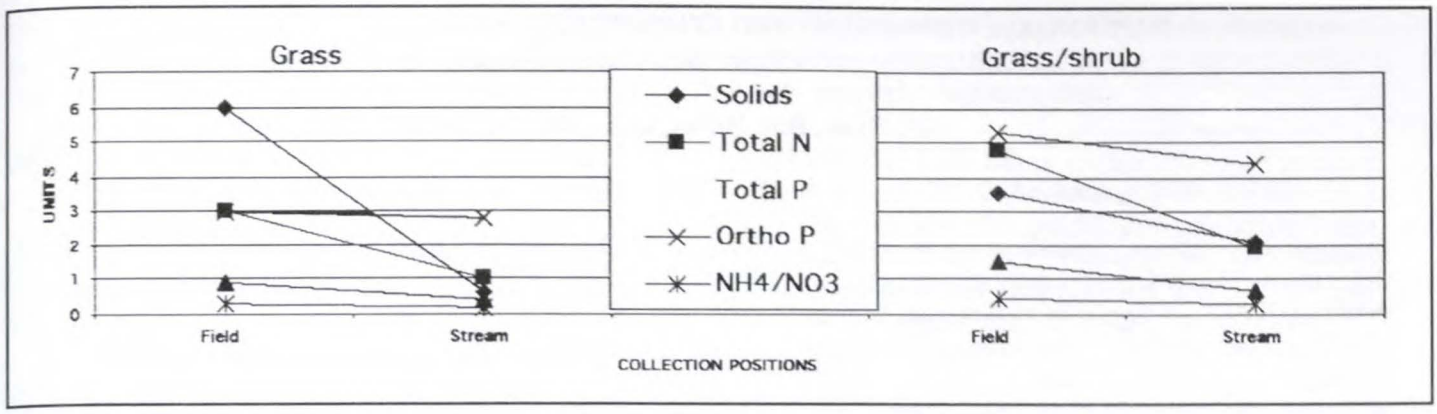

Figure 1.—*Average concentration of pollutants (ppm, relative) during four runoff events in 2001 through a grass buffer and a grass/shrub buffer.*

field and passed through the riparian buffers. We analyzed the water samples for various pollutants including total nitrogen, ortho and total phosphorous, and suspended solids in the KSU Agronomy Department laboratory following appropriate EPA protocols. Two additional sites to include a native woodland and a grassland will be established in subsequent years.

During four runoff events, both suspended solids and total nitrogen were greatly reduced in both the grass only and grass/shrub buffers as water moved downhill (fig. 1). Smaller reductions were found for ammonium/nitrate and total phosphorus. There has been a similar lowering of the pollutants in both the grass and grass/shrub buffers. Thus, initial results show that either type of permanent vegetation is effective in reducing sediment borne or soluble nutrients.

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DEFINING PERENNIAL, INTERMITTENT AND EPHEMERAL CHANNELS IN EASTERN KENTUCKY: APPLICATION TO FORESTRY BEST MANAGEMENT PRACTICES

J.R. Svec, R.K. Kolka, and J.W. Stringer[1]

In Kentucky stream classification is used to determine which forestry best management practice (BMP) to apply in riparian zones. Kentucky defines stream classes as follows (Stringer and others 1998):

a) perennial streams that hold water throughout the year,
b) intermittent streams that hold water during wet portions of the year, and
c) ephemeral channels that hold water only during and immediately after rain events.

Based on flow duration, these definitions of stream class do not represent the natural variation found in the field. The professional literature shows further discrepancies in stream class definitions based on flow duration (Hewlett 1982). Hedman and Osterkamp (1982) defined perennial streams as those having measurable discharge 80 percent of the time, intermittent 10 to 80 percent of the time, and ephemeral <10 percent of the time while Hewlett (1982) defined perennial streams as having water present ≥90 percent of the time.

Generally, as flow duration increases the potential for nonpoint source pollution also increases and BMPs intensify to protect water quality. In Kentucky, the presence, width, and amount of disturbance allowed within the riparian zone, as well as the distance to major soil disturbances (e.g., roads) are all dependent on streamflow duration. Incorrectly determining stream class leads to either increased nonpoint source pollution from increased activity where flow duration would dictate less disturbance, or costly deferral of revenue from too stringent application of the BMPs.

The common practice for determining stream class is the use of USGS Quadrangle topographic maps. Solid blue lines are considered perennial, dotted lines are considered intermittent and channels not defined on the map are considered ephemeral. Although the USGS monitors thousands of perennial streams, they seldom monitor intermittent or ephemeral streams. The map delineation between perennial-intermittent and intermittent-ephemeral is based on conceptual landscape relationships with very little supportive data, and the accuracy is questionable at the site level. Inaccuracies can be seen when measured flow duration data is compared with the USGS map classification.

An alternative to the above method is to measure stream and watershed properties and develop predictive models that relate streamflow duration to these properties. The objective of this study is to develop these predictive relationships. Our ultimate goal is to provide some easily measurable parameters to forest managers and operators so that they can more appropriately apply forestry BMPs in riparian zones.

Twenty-four streams in the Eastern Coal Field Physiographic province of Kentucky were monitored with semi-continuous stage height recording wells to determine the duration of flow from August 2000 to November 2001. Twelve sites were selected in the University of Kentucky's Robinson Experimental Forest. Several sites in the forest are monitored via weirs and flumes and continuous stage height recorders. Six of these sites were chosen for the study. Six additional sites were chosen within the Clemons Fork watershed on the Forest.

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The sites were chosen to represent a range of flow duration. The Clemons Fork Watershed drains slightly more than 3,400 acres in Breathit County, KY. All 12 sites within the Robinson Experimental Forest are within the Clemons Fork watershed. The mapped classification of the Robinson Experimental Forest sites was eight ephemeral and four perennial sites. The 12 additional sites were scattered throughout eastern Kentucky. Candidates for stream sites were selected at random from USGS quadrangle maps falling completely within the boundaries of the Daniel Boone National Forest. Over 100 sites were specified. A number of randomly chosen sites from each category of stream classification (ephemeral, intermittent, or perennial) were selected and visited.

Considerations for selecting a site were remoteness (avoidance of vandalism), access (travel to sites through public land), terrain (negotiable hill slopes; clear stream reaches for measurement), and drainage (one site should not influence the amount of water passing another site). Of the 11 sites chosen, four sites were mapped as ephemeral, five sites were intermittent, and two sites were perennial. One additional site with an ephemeral classification was selected in Blanton Forest in coordination with another ongoing project.

Stream physical parameters measured included bankfull width, bankfull depth profiles, width:depth ratio, flood-prone width, streambed slope, sinuosity, and depth to bedrock (Rosgen 1996). Watershed parameters collected were drainage area, hillslope percent, and occurrences of disturbed area. Stepwise multiple regression models were developed to determine the physical parameters that have the greatest capability in predicting flow duration.

As expected, a range of flow duration and stream and watershed physical properties were found. Initial results suggest that width, streambed slope, and flood prone width are all related to streamflow duration.

Our research provides a field-based method for classifying streams on site. The developed relationships lessen the confusion resulting from varying definitions of the perennial, intermittent, and ephemeral stream classifications. In addition, these relationships provide forest operators easily measurable parameters with which to define stream types in eastern Kentucky. Our research will lead to improved application of Kentucky's forestry BMP guidelines addressing timber harvesting near streams, assuring that riparian zone BMPs are properly implemented.

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PATTERNS IN SPECIES COMPOSITION AND DIVERSITY ALONG INTERMITTENT CREEKS IN THE MISSOURI OZARKS

Cindy E. Becker and Stephen G. Pallardy[1]

The southeast Missouri Ozarks is a rugged, deeply dissected landscape. Intermittent creeks are commonly found throughout the region, yet our understanding of this ecosystem component is poor. Landform features, flooding frequency, and flooding duration are variables known to affect vegetation distribution patterns along perennial systems. We investigated if these variables, and/or the geological substrate and parent material, could also be used to explain distribution patterns of natural communities and species groups along intermittent creeks in the southeast Missouri Ozarks.

We selected 13 intermittent creeks within the watersheds of the Current, Jacks Fork, and Eleven Point Rivers for sampling: 11 1st- and 2nd-order creeks, one 3rd-order, and one 5th-order creek. For each creek, we collected information at the watershed level, e.g., underlying geological strata, drainage area, stream order, and stream length. Eight to ten 20-m by 20-m plots were located down the length of each creek within the bounds of a distinct fluvial landform. For each plot we recorded waterway width, elevation above creek channel, geological parent material, percent surficial rock, soil textural, and soil structural characteristics, as well as collected overstory, understory, and ground flora data. All woody stems >1.25 cm d.b.h. were recorded to species and d.b.h. and all ground flora vegetation (<1 m-tall) was recorded to species and given a cover class code. Detrended Correspondence Analysis (DCA) ordination of all strata was conducted. DCA sample axis scores were examined for relationships with environmental variables using Spearman rank correlation analysis and analysis of variance to identify potential classifications of vegetation based on physical variables. From these results a multi-factor, physically-based, hierarchical classification incorporating highly correlated environmental variables was developed. Plots distributed to these classes were tested for within-class homogeneity and between-class differences with the Multi-Response-Permutation Procedure (MRPP).

We found that fluvial landform parameters, e.g., soil texture, soil development, and their associated characteristics, such as flooding intensity, duration, and frequency, were important predictors of species distribution patterns in higher order creeks (3rd and 5th). Differential suites of species were associated with channel, active floodplain, and terrace features. These features differed in their elevation above the creek channel, and soil textural characteristics in both the surface and subsurface horizons. Channel features, unique to intermittent high order creeks, had two distinct vegetation strata, an extremely dense shrub layer of the flood-tolerant species, *Hamamelis vernalis* Sarg. and *Ptelea trifoliata* L. and a sparse scattering of ground flora species adapted to intense, frequent flood disturbance. Active floodplains were typically three-tiered, with a dense closed canopy of mesic overstory species, *Quercus muehlenbergii* Engelm., *Q. alba* L., and *Fraxinus americana* L., shading an understory of *Lindera benzoin* (L.) Blume, *Asimina triloba* (L.) Dunal, and *F. americana* saplings and a lush ground flora layer of forbs and woodland grasses. Terrace features had relatively open mature stands of *Q. alba* with *Cornus florida* L., low-growing shrubs, woody vines, and woody seedlings in the understory.

For lower order intermittent creeks (1st - 2nd), soil texture, drainage and base-saturation, and vegetation were more strongly associated with parent material and underlying bedrock type than with fluvial landform. In general, individual lower order intermittent creeks were characterized by similar parent material and

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underlying geologic strata along their length. Creeks developed in Roubidoux-Upper Gasconade cherty limestone and sandstone substrates with rocky, coarse-textured soils, were characterized by an open, three-tiered mixed *Quercus-Carya/Cornus* dry-mesic forest and a sparse ground flora dominated by upland generalist species such as *Parthenocissus quinquefolia* (L.) Planch., *Desmodium nudiflorum* (L.) DC., *Smilacina racemosa* (L.) Desf., and *Sanicula canadensis* L. and species of upland mesic/alfic forests (*Cimifuga racemosa* (L.) Nutt., *Geranium maculatum* L., *Agrimonia rostellata* Wallr., and *Brachyelytrum erectum* (Schreb.) Beauv.).

Conversely, creeks developed in lower Gasconade and Eminence dolomitic limestone substrates, with relatively chert-free, fine-textured soils, were characterized by a shaded, multiple-tiered mixed hardwood mesic bottomland forest community (*Carya cordiformis* (Wangenh.) K. Koch-*Q. muehlenbergii-Ulmus rubra* Muhl./*Lindera benzoin-Asimina triloba*), and an extremely lush, diverse ground flora. Further, within intermittent low-order creeks of like-bedrock types, patterns of water flow along the length of a creek were also strongly related to species distribution patterns. For all vegetation strata, species composition and structure changed in a downstream direction (e.g., increases in richness in the ground flora and understory, and increases in total cover and density in the ground flora and understory, respectively). These changes in species diversity, abundance, and cover were associated with position within a waterway, local watershed characteristics, and water flow.

From Indicator Species Analysis, indicator species groups were identified. Seven ecological species groups comprised of co-occurring ground flora and understory species were identified and described. Species and species groups were not unique to a community, but rather the dominance of a species or group varied across communities. Likely, high inherent ecological species amplitude and substantial micro-habitat variability created from deposition, erosion, and scouring of flood events, treefall pit and mounds, and bedrock and parent material complexity allow wide distribution of Ozark species.

Intermittent creeks were highly floristically and physically diverse, complex components of the Ozark landscape. Patterns in diversity and differences in species composition were strongly related to differences in the underlying geology and parent materials along lower-order intermittent creeks, and to landform parameters along higher-order intermittent creeks.

GROUND-FLORA COMMUNTIES OF HEADWATER RIPARIAN AREAS IN AN OLD-GROWTH CENTRAL HARDWOOD FOREST

P. Charles Goebel, David M. Hix, Clayton E. Dygert, and Kathryn L. Holmes[1]

ABSTRACT.—The composition and structure of ground-flora vegetation was examined across headwater riparian areas of Johnson Woods, an old-growth forest located in north-central Ohio. While the distribution patterns of these species groups is variable, classification and gradient analyses indicate that ground-flora vegetation is related strongly to landform and distance from the bankfull channel, suggesting that species are ordered along a complex environment from the stream edge, across the floodplains, and into the adjacent uplands. No significant differences in species richness were observed between the floodplain and upland landforms. However, there are differences in functional lifeform guilds between landforms. Specifically, graminoids, annual forbs, and perennial forbs dominate the floodplains, while woody seedling and vines dominate the adjacent uplands.

Our understanding of riparian areas has evolved considerably over the past 20 years from viewing riparian areas as individual units of the landscape to viewing them as ecotones located between aquatic and terrestrial systems (Gregory and others 1991, Illhardt and others 2000). Through a variety of empirical studies, we have discovered that riparian areas are complex ecotones characterized by gradients of structural and functional change from the water's edge to the uplands, and it is the juxtaposition of these dynamic ecotones with water which leads ultimately to many of the important ecological services provided by riparian areas (Gregory and others 1991, Illhardt and others 2000, Naiman and others 2000). For example, surface and subsurface connections with water typically result in a diverse array of different habitat that supports numerous plant and animal species (Pollack and others 1998, Brinson and Verhoeven 1999). Furthermore, these ecotones are known to regulate inputs of organic matter to the aquatic systems (Vannote and others 1980), and consequently, control the structure of the aquatic foodweb, as well as the cycling of nutrients at the watershed level (Peterson and others 2001).

Based on the important ecological contributions riparian areas provide, understanding the natural variability and patterns in riparian vegetation, as well as the factors that mediate the structure and function of riparian vegetation, is critical for predicting vegetation change in these complex systems. In most riparian settings, landscape features resulting from the interaction of hydrologic and geomorphic processes are believed to be important constraints on the composition and structure of riparian areas (Gregory and others 1991, Auble and others 1994, Hupp and Osterkamp 1996). While the distribution of woody vegetation across riparian areas is closely tied to hydrogeomorphic processes, the composition and spatial distribution of ground flora species can also be influenced strongly by a variety of additional factors, including the composition and structure of other vegetation layers (McCune and Antos 1981, Goebel and Hix 1997), microtopography (Titus 1990), and seasonal change (Goebel and others 1999).

In order to understand which of these factors are the most important variables controlling the distribution of ground-flora vegetation, it is

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important that we determine the factors that influence vegetation patterns across riparian areas. One common approach is to examine how individual species or groups of species are arranged on different fluvial landforms, as fluvial landforms are surrogates of the prevalent hydrogeomorphic processes of the stream valley (Fetherston and others 1995). It is important to understand that this approach is based on the premise that species with similar distributions along environmental gradients (i.e., a riparian ecotone) can be grouped together to help elucidate the relationships between vegetation and the environment (Pregitzer and Barnes 1982). Additionally, in areas where the factors influencing the composition, structure, and diversity of ground-flora vegetation are dynamic and spatially heterogeneous (such as an individual landform or riparian ecotone), the use of species groups or functional plant guilds, may be more useful for elucidating the ecological patterns and underlying ecosystem processes than individual species alone (Pabst and Spies 1998).

Because vegetative descriptions of headwater Central Hardwood riparian areas are lacking, and characterization of the composition and structure of riparian vegetation provides the foundation for subsequent ecological and management studies, we examined the relationships among ground-flora vegetation and environmental factors across riparian ecotones located in a small old-growth headwater basin located in north-central Ohio. Specifically, we addressed the following questions:

1) How does ground flora vegetation change across riparian areas in this old-growth headwater basin?
2) Are there individual or groups of ground-flora species that can be used to help identify the extent of headwater riparian area?

STUDY AREA

We conducted our study at the Johnson Woods State Nature Preserve (JWSNP), a virgin 83 ha old-growth forest located in Wayne County, Ohio (fig. 1). Johnson Woods is located in the Western Glaciated Allegheny Plateau, a maturely dissected plateau modified by glaciation and characterized by rounded hills, ridges, broad valleys, and a variety of glacial landforms (Keys and others 1994). The study area has a humid-continental climatic regime with mean annual temperatures of 10°C. Mean annual precipitation ranges from 900 to 1,020 mm, and is fairly evenly distributed throughout the year (Keys and others 1994).

The uplands of Johnson Woods are dominated by gentle slopes (< 5 percent), well-drained soils, and old-growth plant communities of mixed oaks (*Quercus* spp.), hickories (*Carya* spp.), and an increasing component of shade-tolerant species, including sugar maple (*Acer saccharum* Marsh.) and American beech (*Fagus grandifolia* Ehrh.). In addition to these well-drained uplands, there are many depressional areas or vernal pools that feed a first-order, intermittent stream characterized predominately by narrow floodplains and a gentle slope into the uplands (fig. 1). The poorly drained soils associated with these vernal pools are often deep and underlain by glacial outwash and are dominated by a mixture of woody species, including swamp white oak (*Quercus bicolor* Willd.), white ash (*Fraxinus americana* L.), and buttonbush (*Cephalanthus occidentalis* L.).

METHODS

During the summer of 2001, we collected ground-flora data from 24 transects that bisected the stream valley along the entire 0.75 km length of the intermittent channel located in JWSNP. Each transect extended from the stream edge (i.e., bankfull channel edge) into the uplands, with transects arrayed perpendicular to streamflow. The location of the first transect was determined randomly beginning 40 m from the preserve boundary; successive transects were then located randomly at least 20 m but no more than 40 m apart in the upstream direction. Sample plots were located along each transect systematically at 5 m intervals (e.g., 0, 5, 10, 15, and so on), with at least two sample plots located in the adjacent upland forest on each transect. Individual transects had between 6 to 10 plots, depending on the valley width, for a total of 193 sample plots.

In each sample plot, we recorded the landform (floodplain, upland), distance from the stream bankfull channel, and percent cover of all species < 1 m tall in a 1 m^2 quadrant. Cover classes included: < 1 percent, 1-5 percent, 6-10 percent, 11-20 percent, 21-40 percent, 41-70 percent, and 71-100 percent. We also sorted the species into functional lifeform groups (annual forbs, perennial forbs, graminoids, pteridophytes, epiphytes, woody vines, woody shrubs, and woody seedlings) and estimated the percent cover of each group. Nomenclature and life form categories follow Voss (1996), except for pteridophytes (Gleason and Cronquist 1991).

Finally, we classified each species in terms of its wetland indicator status—obligate wetland

(OBL), facultative wetland (FACW), facultative (FAC), facultative upland (FACU), and obligate upland (UPL) (Reed 1988). By definition: obligate wetland species (OBL) are plants that almost always occur (99 percent) in wetlands; facultative wetland species (FACW) are plants that usually occur (67-99 percent) in wetlands; facultative species (FAC) are plants that occur (33-67 percent) in both wetlands and uplands; facultative upland species (FACU) are plants that occasionally occur (1-33 percent) in wetlands; and, obligate upland species (UPL) are plants that almost always occur (99 percent) in the uplands.

We used several different methods to quantify and compare patterns of change in ground-flora vegetation and physiographic factors. Because rare species can mask patterns in vegetation, we dropped species occurring on less than 5 percent of all sample plots (9 out of 193 plots) for all classification and ordination analyses. An initial classification based on the floristic dissimilarity of species on all plots was made using Two Way Indicator Species Analysis (TWINSPAN) (Hill 1979) using default settings and seven pseudospecies cut levels corresponding to the lower limit of each percent cover class, e.g., 0, 1, 6, 11, 21, 41, and 71. The results of this initial classification provided the basis for eight ecological species groups, each named for the species with the greatest ecological amplitude and importance. Mean cover of each ecological species group was calculated by landform and distribution patterns examined.

Distributions of individual species were examined with detrended correspondence analysis (DCA) using PC-ORD software (McCune and Mefford 1995). We used Spearman rank correlations of the sample unit scores and environmental factors to help interpret the relationships between the first two DCA axes. We also used a one-way analysis of variance (ANOVA) to compare the species richness (total species per m^2), percent cover of lifeform categories between the floodplain and upland landforms, and the total cover of wetland (OBL and FACW) and upland (FAC, FACU, and UPL) species between the floodplain and upland landforms. Data were transformed (arcsin transformation for percentage data) prior to analyses to stabilize variances (Zar 1996) and all analyses were conducted using SAS version 8.1 (SAS Institute, Cary, NC).

Finally, we supplemented the traditional gradient analyses with Dufrene and Legendre's (1997) indicator analysis using PC-ORD (McCune and Mefford 1995). These analyses

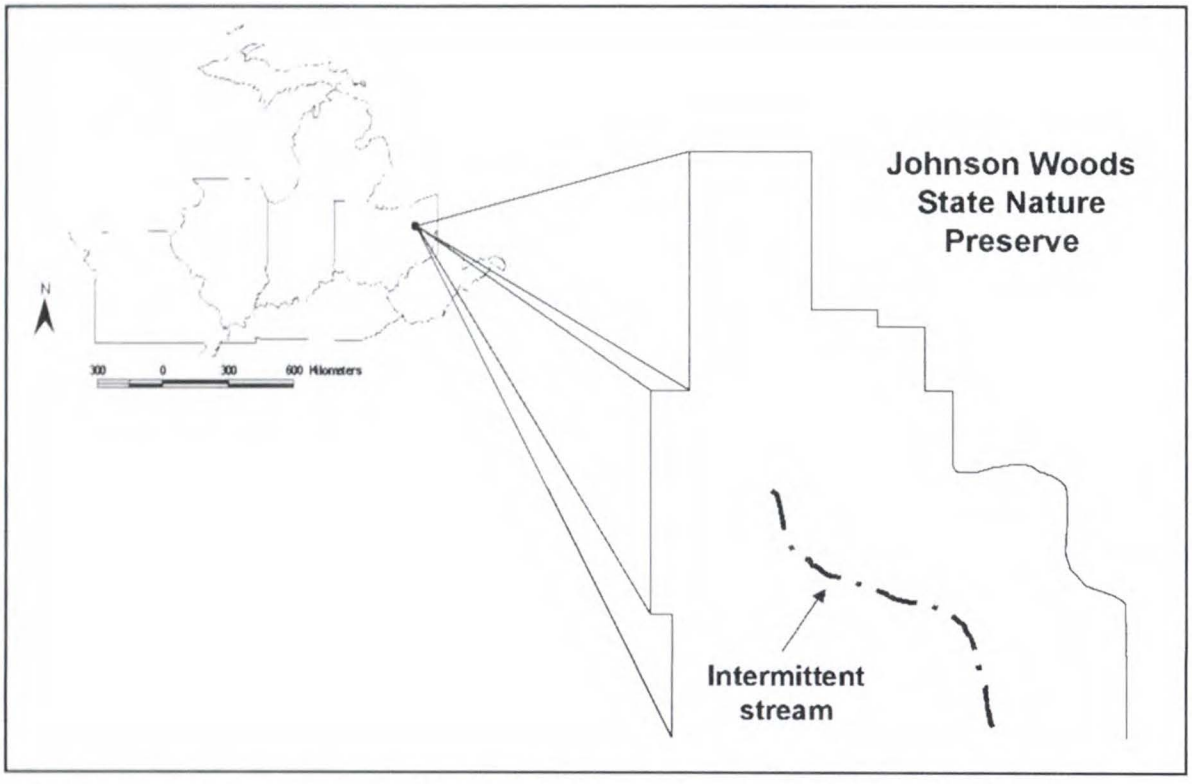

Figure 1.—*Location of Johnson Woods State Nature Preserve, an old-growth forest located in north-central Ohio.*

use Monte Carlo permutation procedures to test the association of each species with each valley landform, and generate a *p*-value that is the proportion of randomized trials in the permutation procedure with an indicator value equal to or exceeding the observed indicator value (Dufrene and Legendre 1997).

RESULTS

Ground-flora Classification and Ordination

We sampled a total of 75 species, of which almost half of these species are perennial forbs (45 percent), while the remaining species are classified as woody seedlings (20 percent), woody shrubs (8 percent), pteridophytes (8 percent), woody vines (7 percent), graminoids (5 percent), annual forbs (5 percent), or epiphytes (1 percent). Approximately a third of all species sampled are considered wetland species (classified as either facultative wetland (28 percent) or obligate wetland (3 percent) species), while the remaining are classified as upland species (facultative (20 percent), facultative upland (32 percent), or upland obligate (4 percent) species). According to the National Wetland Indicator List, 10 of the 75 species sampled (13 percent) are not classified (Reed 1988), but most are typically found in moist or wet forest sites (and therefore were classified as facultative species for the remaining analyses).

Using TWINSPAN, we classified the 28 most common species (those occurring on at least 5 percent of the sample plots) into eight groups within which the component ground-flora species have similar patterns of occurrence. These include the following species groups: *Viburnum*, *Acer*, *Parthenocissus*, *Fraxinus*, *Quercus*, *Circea*, *Impatiens*, and *Leersia* (table 1). The occurrence of eight different species groups reflects the variability in species composition and dominance within and among the different landforms. Three of the species groups are common on the floodplains (*Impatiens*, *Leersia*, and *Fraxinus*); all of which are dominated primarily by either facultative wetland or obligate wetland species (e.g., *Leersia virginica* Willd.) (see Appendix).

The upland landforms were dominated by four groups of ground-flora species, including members of the *Acer*, *Fraxinus*, *Parthenocissus*, and *Impatiens* groups (table 1). Only the *Acer* group is dominated entirely by facultative or facultative upland species (sugar maple, American beech, partridge-berry (*Mitchella repens* L.), Solomon-seal (*Polygonatum pubescens* (Willd.) Pursh), and black cherry (*Prunus serotina* Ehrh.). Members of the other species groups (e.g., *Viburnum*, *Circea*, *Quercus*) are less dominant across individual riparian ecotones.

The ordination analysis of the ground-flora data demonstrates the complex compositional gradient that occurs along riparian ecotones of Johnson Woods (fig. 2). The first DCA axis (eigenvalue = 0.49) represents a compositional and topographical gradient from the uplands to the streamside, with the extremes dominated by members of the *Acer* group in the uplands, and the *Leersia* and *Impatiens* species groups strongly associated with the floodplains. Although there is some overlap among sample points, the distribution of plots is arranged by landform. Correspondingly, the moderate Spearman rank correlation between DCA first axis scores and distance from the stream channel (m) supports this interpretation ($r = -0.39$; $P < 0.05$).

The distribution of sample points and species along the second DCA axis (eigenvalue = 0.25) is less clear. The gradient is determined largely by the difference between the *Viburnum* species group, with high axis 2 scores, and the *Parthenocissus* and *Fraxinus* groups, both with low axis 2 scores (fig. 2). This axis may represent a gradient of soil moisture, as several of the species that comprise both the *Parthenocissus* and *Fraxinus* groups are either facultative wetland or obligate wetland species. However, further research related to the soils of each plot is needed to determine whether, in fact, this ordination axis does represent a gradient of soil moisture nested within the gradient of

Table 1.—*Mean cover of ground-flora ecological species groups by landform for riparian areas of Johnson Woods, north-central Ohio. See Appendix for list of species in each group.*

Species group	LANDFORM CATEGORY	
	Floodplain (*n*=56)	Upland (*n*=137)
Viburnum	0.72 (0.45)[a]	2.88 (0.88)
Acer	2.11 (0.98)	10.95 (1.29)
Parthenocissus	3.04 (0.96)	8.71 (1.13)
Fraxinus	5.24 (1.42)	10.35 (1.28)
Quercus	1.02 (0.20)	0.62 (0.11)
Circea	1.74 (0.68)	2.34 (0.43)
Impatiens	23.27 (4.00)	6.61 (1.30)
Leersia	6.79 (1.54)	0.31 (0.23)

[a]Values are mean ± 1 standard error.

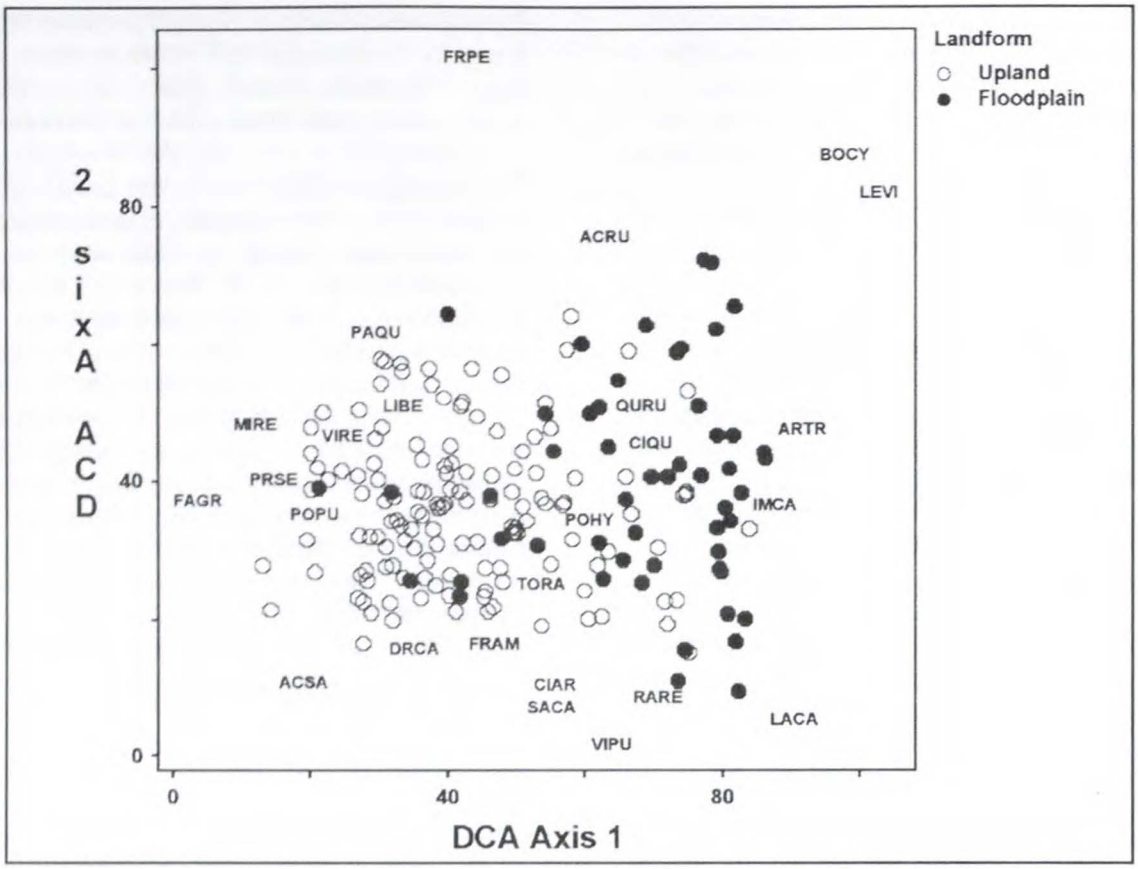

Figure 2.—*DCA ordination of the ground-flora of Johnson Woods, showing the distribution of sample plots by landform and common (those occurring on > 5 percent of the sample plots) ground-flora species. See Appendix for species codes.*

increasing distance from the stream channel as represented by the first DCA axis.

Species Richness and Ground-Flora Cover Analyses

No significant difference in species richness (per m^2) was observed between the floodplain and upland landforms (table 2). Similar results are observed when total cover is examined, with a total cover of 53.18 percent $\pm$ 5.74 percent (mean + 1 SE) for the floodplain and 49.50 percent $\pm$ 3.18 percent for the upland landforms, respectively. However, when the ground-flora was separated into different lifeform categories there are significant differences between the floodplain and upland landforms. For instance, the floodplains have significantly higher cover of annual forbs, perennial forbs, and graminoids than the adjacent upland landforms. Conversely, the uplands have higher cover of woody seedlings and vines. Finally, the upland landforms are dominated by species classified as upland indicator species (FAC, FACU, UPL), while the floodplains are dominated by both upland and wetland indicator species (FACW, OBL).

Ground-Flora Indicator Analyses

Using Dufrene and Legendre's (1997) indicator analysis, we identified 11 common species that are strong indicators of either the upland or floodplain landforms, and by extension can be used to help identify the extent of the riparian area. Seven of the species are upland indicator species, and include a variety of perennial forbs, woody seedlings, and woody vines: sugar maple, jack-in-the-pulpit, (*Arisaema triphyllum* (L.) Schott), American beech, white ash (*Fraxinus americana* L.), Virginia creeper (*Parthenocissus quinquefolia* (L.) Planchon), Solomon-seal, and black cherry (table 3). The remaining four species are significant floodplain indicators and include: false nettle (*Boehmeria cylindrica* (L.) Sw.), touch-me-not (*Impatiens capensis* Meerb.), wood-nettle (*Laportea canadensis* (L.) Wedd.), and white grass (*Leersia virginica* Willd.).

DISCUSSION

Unlike other regions of North America, little research has focused on understanding compositional and structural changes across riparian areas of Ohio, or the Central Hardwood region. Those studies that have examined riparian

Table 2.—*Comparison of species richness, percent cover by lifeform, and percent cover by wetland indicator status (upland vs. wetland) between floodplain and upland landforms for riparian areas of Johnson Woods, north-central Ohio*

Parameter	LANDFORM CATEGORY		ONE-WAY ANOVA	
	Floodplain (*n*=56)	Upland (*n*=137)	F-statistic	*P*-value
Richness (species per m2)	5.02 (0.35)[a]	5.27 (0.20)	0.42	0.517
Total cover (%)	53.18 (5.74)	49.50 (3.18)	0.36	0.552
Lifeforms				
Cover of annual forbs (%)	12.16 (2.67)	4.97 (0.96)	9.96	0.002
Cover of perennial forbs (%)	22.77 (0.96)	10.55 (1.40)	17.94	<0.001
Cover of graminoids (%)	8.13 (1.66)	2.14 (0.60)	15.88	<0.001
Cover of pteridophytes (%)	0.14 (0.14)	1.16 (0.45)	2.09	0.150
Cover of woody shrubs (%)	0.43 (0.24)	2.30 (0.61)	3.75	0.054
Cover of woody seedlings (%)	4.16 (1.00)	14.68 (1.49)	18.78	<0.001
Cover of woody vines (%)	5.38 (1.27)	13.60 (1.49)	11.05	0.001
Wetland Indicators				
Cover of upland species (%)	26.70 (3.33)	36.12 (2.43)	4.68	0.032
Cover of wetland species (%)	24.98 (3.88)	10.70 (1.54)	17.08	<0.001

[a]Values are mean ± 1 standard error.

areas in the region have usually focused on either a single landform, such as floodplains (e.g., Williams and others 1999), or individual streamside forests (McCarthy and others 1987, Hix and Pearcy 1997). None of these studies have treated riparian areas as ecotones in an attempt to understand how vegetation patterns may change across these small-scale gradients, especially in headwater stream systems. Consequently, our research is one of the first quantitative studies to

1) focus on riparian areas as ecotones in the region, and
2) to quantify vegetation change across these riparian ecotones.

Our results show that, as in many other riparian landscapes, ground-flora vegetation patterns across riparian areas of Johnson Woods are variable yet ordered along a complex compositional gradient related to distance from the bankfull channel (Franz and Bazzaz 1977, Pabst and Spies 1998, Goebel 2001). Such relationships support the assertion that riparian areas are ecotones rather than discrete units of the landscape, as the gradient is influenced by a host of interacting factors, including physiography, soils, disturbances, and climatic conditions (Gosz 1993). Specifically, the factors believed to have the greatest influence on riparian vegetation are:

1) hillslope processes,
2) hydrologic disturbances,
3) tolerance of saturated soils, and
4) mineral soil disturbances (Gregory and others 1991, Pabst and Spies 1998, Naiman and others 2000).

Although more research related to the specific mechanisms controlling the changes in vegetation across these riparian areas is needed, it is likely that a tolerance for saturated soils and seasonal hydrologic disturbances (as reflected by the dominance of wetland species on the floodplain landforms) are the most important factors regulating the distribution of ground flora species in the gently sloping headwater systems of Johnson Woods.

Contrasted to other riparian areas of the Upper Midwest, the headwater systems of Johnson Woods are similar in terms of species richness to old-growth headwater systems located in Upper Michigan. In these first-order stream valleys dominated by northern hardwood forests, Goebel (2001) found no significant differences in species richness between floodplains and upland landforms (mean richness = 5.4 and 4.0, respectively). The similarities in species richness between the two landforms may be the result of a variety of microsites that support a diverse array of species in both the uplands as well as the riparian areas. These similarities,

Table 3.—*Indicator analysis of common ground-flora species of riparian ecotones in Johnson Woods. Comparison is between floodplain and upland landform classes*

Species name	Code	Observed indicator value	Indicator value from randomized permutation procedure[a]	P[b]
Acer rubrum	ACRU	19.5	17.6 (2.6)	0.197
Acer saccharum	ACSA	32.3	20.4 (3.0)	.003†
Arisema triphyllum	ARTR	14.5	6.8 (1.9)	.007†
Boehmeria cylindrica	BOCY	18.9	5.1 (1.7)	.001*
Cinna arundinacea	CIAR	9.0	9.5 (2.2)	.503
Circea quadrisulcata	CIQU	9.5	9.5 (2.2)	.396
Dryopteris carthusiana	DRCA	5.1	4.3 (1.5)	.273
Fagus grandifolia	FAGR	24.0	12.9 (2.4)	.001†
Fraxinus americana	FRAM	30.4	23.7 (3.1)	.037†
F. pennsylvanica	FRPE	6.6	6.2 (1.7)	.352
Impatiens capensis	IMCA	36.2	18.0 (2.6)	.001*
Laportea canadensis	LACA	37.4	15.6 (2.6)	.001*
Leersia virginica	LEVI	49.6	11.1 (2.2)	.001*
Lindera benzoin	LIBE	6.2	5.7 (1.8)	.311
Mitchells repens	MIRE	3.8	4.3 (1.6)	.623
Parthenocissuss quinquefolia	PAQU	33.2	21.1 (2.8)	.003†
Polygonum hydropiper	POHY	10.0	9.7 (2.2)	.364
Polygonatum pubescens	POPU	23.6	12.7 (2.5)	.002†
Prunus serotina	PRSE	16.4	9.8 (2.2)	.015†
Quercus rubra	QURU	6.7	4.0 (1.4)	.080
Ranunculus recurvatus	RARE	2.7	4.3 (1.5)	.999
Sambucus canadensis	SACA	4.7	5.2 (1.7)	.537
Toxicodendron radicans	TORA	24.7	22.2 (2.8)	.169
Viola pubescens	VIPU	15.2	12.9 (2.4)	.150
Viburnum recognitum	VIRE	5.3	4.3 (1.5)	.264

[a] Mean (± 1 standard deviation).
[b] Proportion of randomized trials with indicator value equal to or exceeding the observed indicator value;
* indicates a floodplain indicator species,
† indicates an upland indicator species.

however, are not observed when second-growth sites are examined. For example, in headwater systems located in both unglaciated western Pennsylvania and north-central Minnesota, the ground-flora of the floodplains have higher ground-flora species richness than the adjacent uplands (Williams and others 1999, Goebel 2001).

While species richness of headwater systems in north-central Ohio and Upper Michigan are similar, the structure of the ground-flora vegetation is markedly different. In Johnson Woods we observed riparian areas dominated by forbs and graminoids, while riparian areas in Upper Michigan are dominated almost exclusively by graminoids (Goebel 2001). These differences are most likely attributed to the different types of hydrogeomorphic processes operating in each system. For example, the influence of flooding on the riparian forest canopy of Johnson Woods may be relatively benign, as the forest canopy is relatively continuous across the riparian area into the uplands. Conversely, the forest canopy of headwater riparian areas in Upper Michigan is more heterogeneous, grading from a closed canopy of sugar maple and eastern hemlock (*Tsuga canadensis* L.) to open sedge (*Carex* spp.) meadows influenced strongly by seasonal flows and active beaver dams (Goebel 2001).

The fact that distance from the bankfull channel was only moderately correlated with the DCA axes suggests that distance from the bankfull channel by itself is not a reliable indicator of riparian status. However, an approach based on landform classification and indicator species has promise to help identify the extent of the

riparian area. Our results suggest that a mixture of different species assemblages may be a good indicator of the extent of riparian areas in headwater systems of north-central Ohio. Specifically, the presence of several graminoid and forb species can be used to help determine the extent of the floodplain in these headwater systems, a process that can be problematic considering the gentle slopes and little stream valley development associated with these headwater systems. Additionally, the presence of white grass, touch-me-nots, wood-nettle, and false nettle likely indicate a floodplain surface and a riparian environment. Such information may be useful as riparian area management moves from a "one-size-fits-all" approach to a more functional approach that takes into consideration a variety of ecological services provided by riparian areas, many of which are mediated by specific hydrogeomorphic processes (Illhardt and others 2000).

Finally, while descriptive, the information reported here provides a basis for future research directives focused on understanding the functional processes associated with these critical headwater systems. By conducting these analyses in the few remaining intact old-growth forests located in the Central Hardwood region such as Johnson Woods, these areas provide an excellent opportunity to examine the influences of different hydrogeomorphic processes on riparian vegetation in an undisturbed setting. While restoring all the functional properties associated with these riparian areas in disturbed headwater systems of north-central Ohio may be unrealistic, it is only through understanding the dynamics of riparian areas in an undisturbed setting that we can decompose the factors that control the properties of specific riparian areas. Consequently, these old-growth riparian areas of Johnson Woods provide an "endpoint" along a disturbance gradient with which we can compare the composition, structure, and function of manipulated or restored riparian areas in north-central Ohio and beyond.

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APPENDIX

List of species in each ground-flora ecological species group, including species code (e.g., FRPE), lifeform class (e.g., annual forb, perennial forb, graminoids, pteridophytes, woody seedling, woody shrub, woody vine), and wetland indicator status (e.g., OBL, obligate wetland; FACW, facultative wetland; FAC, facultative; FACU, facultative upland; UPL, obligate upland).

***Viburnum* species group**

Fraxinus pennsylvanica Marshall – FRPE; woody seedling; FACWa
Lindera benzoin (L.) Blume – LIBE; woody shrub; FACW
Viburnum recognitum Fernald – VIRE; woody shrub; FAC

***Acer* species group**

Acer saccharum Marsh. – ACSA; woody seedling; FACU
Fagus grandifolia Ehrh. – FAGR; woody seedling; FACU
Mitchella repens L. – MIRE; perennial forb; FACU
Polygonatum pubescens (Willd.) Pursh – POPU; perennial forb; FAC
Prunus serotina Ehrh. – PRSE; woody seedling; FACU

***Parthenocissus* species group**

Cinna arundinacea L. – CIAR; graminoid; FACW
Dryopteris carthusiana (Villars) H.P.Fuchs – DRCA; pteridophyte; FAC
Parthenocissuss quinquefolia (L.) Planchon – PAQU; woody vine; FACU

***Fraxinus* species group**

Fraxinus americana L. – FRAM; woody seedling; FACU
Polygonum hydropiper L. – POHY; annual forb; OBL
Sambucus canadensis L. – SACA; woody seedling; FACW
Toxicodendron radicans (L.) Kuntze – TORA; woody vine; FAC

***Quercus* species group**

Acer rubrum L. – ACRU; woody seedling; FAC
Quercus rubra L. – QURU; woody seedling; FACU

***Circea* species group**

Circea quadrisulcata (Maxim.) Franchet & Savat – CIQU; perennial forb; FAC
Ranunculus recurvatus Poiret – RARE; perennial forb; FAC
Viola pubescens Aiton – VIPU; perennial forb; FAC

***Impatiens* species group**

Arisaema triphyllum (L.) Schott – ARTR; perennial forb; FACW
Impatiens capensis Meerb. – IMCA; annual forb; FACW
Laportea canadensis (L.) Wedd. – LACA; perennial forb; FACW

***Leersia* species group**

Leersia virginica Willd. – LEVI; graminoids; OBL
Boehmeria cylindrica (L.) Sw. – BOCY; perennial forb; FACW

COMPOSITION AND STRUCTURE OF AN OLD-GROWTH FLOODPLAIN FOREST OF THE LOWER KASKASKIA RIVER

John B. Taft[1]

ABSTRACT.—Compositional and structural properties of canopy, shrub/sapling, and ground-cover strata were measured within an old-growth floodplain forest bordering the lower Kaskaskia River in southwestern Illinois. Basal area for trees was estimated at 31.8 m^2/ha, tree density was 398 trees/ha with 27 species recorded in the canopy stratum. The dominant tree species were *Acer saccharinum, A. negundo, Carya laciniosa, Ulmus rubra,* and *Quercus macrocarpa.* Presently, there appears to be insufficient recruitment to maintain *A. saccharinum* at current abundance levels in the canopy. Total sapling/shrub density, including woody vines, was estimated to be 7,440 stems/ha and 24 species were recorded. Dominant species were *Toxicodendron radicans, Campsis radicans, Ulmus rubra, Celtis occidentalis,* and *Ilex decidua.* Ground-cover vegetation was characterized by species density (species per 1/4-m^2 quadrat) of 4.88, with 81 species (in 240 quadrats), and 80.8 percent cover. Dominant species were *Laportea canadensis, Toxicodendron radicans, Ranunculus septentrionalis, Viola pratincola,* and *Galium aparine.* The floodplain is a mosaic of intergrading wet, wet-mesic, and mesic floodplain forest natural communities.

An assessment was made of the vegetation within selected portions of the Kaskaskia River Corridor (KRC) with a focus on native plant communities (Taft, unpubl. report A). The KRC is an area of privately owned lands in Clinton, Washington, and St. Clair Counties in southwestern Illinois bordering the Kaskaskia River between Carlyle, and a reservoir at Carlyle, and Fayetteville, IL (Evans and others, unpubl. report). The vegetation of the KRC is a particularly noteworthy resource because it includes some of the largest tracts of unfragmented forest remaining in Illinois, including some old and old second-growth stands of floodplain forest and flatwoods, providing habitat for area-sensitive native plant and animal species. About 343 acres of forest within the KRC are recognized as statewide-significant natural areas (i.e., remain in a state of high ecological integrity (White 1978)) including about 230 acres of flatwoods and 110 acres of floodplain forest. In addition to the sites recognized as natural areas, many other forest stands in the KRC have considerable ecological integrity and/or restoration potential.

The main goals of the vegetational survey were to identify and describe the natural plant communities present in the KRC and provide estimates on composition, abundance, and diversity patterns in an old-growth forest associated with the Kaskaskia River. A specific focus was to estimate the spatial heterogeneity of old-growth floodplain forest community types, to examine patterns of recruitment among canopy species, and to document composition, abundance, and diversity patterns among all vegetation strata in an old-growth floodplain forest complex. These data provide a representative standard for comparison with other floodplain forests in the region.

NATURAL DIVISION AND PHYSIOGRAPHIC REGION

The KRC occurs in the Southern Till Plain Natural Division (Schwegman and others 1973). The lower portion of the Kaskaskia River basin, the area including the KRC, occurs in the Mt. Vernon Hill Country physiographic division (Leighton and others 1948), a region characterized by surface deposits of the Illinoian glacial period. Elevation within the KRC ranges from a

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minimum of about 116 m above sea level at the lowest reach of the Kaskaskia River in the study area at Fayetteville, to a maximum of about 180 m above sea level at the "Pelican Pouch" glacial feature 6 km SSW of Carlyle. However, by far, most of the forest in the study area occurs between the 116 m and 125 m contour intervals. The KRC occurs within the Kaskaskia River Ecosystem Partnership Assessment Area, a 9,248 km^2 area mostly defined by the drainage basin of the Kaskaskia River. The vegetational and physiographic characteristics of that region previously have been described (Taft unpubl. report B).

METHODS

The vegetation in the KRC was examined with two methods:

1) basic floristic inventory of available habitats, and
2) quantitative vegetation sampling in representative forest habitats.

The results of quantitative sampling in a floodplain forest and, in part, adjacent flatwoods, are reported here.

Quantitative Sampling

Overstory, shrub/sapling, and ground-cover sample data were collected in a stratified-random sampling regime with nested subsamples for each vegetation stratum. In order to provide a description of the composition, abundance, and diversity patterns associated with the gradient from forest bordering the river to upland forest habitat, the vegetation was sampled across the floodplain, extending in a distance gradient perpendicular from the Kaskaskia River, until reaching upland forest habitat, in this case flatwoods. The site sampled, Jackson Slough Woods (038° 25′ 23.66″ N latitude, 089° 45′ 52.99″ W longitude), is 181-acre unit within a larger forest and is recognized by the Illinois Natural Areas Inventory as a high-quality natural area with Grades A and B floodplain forest and southern flatwoods (White 1978).

Vegetation was sampled using two parallel transects, each approximately 275 m long, oriented perpendicular to the axis of the Kaskaskia River. The locations of the transects and plots were determined randomly. Trees (woody stems ≥ 6 cm diameter-at-breast-height [dbh]) were sampled in ten 0.05-ha circular plots on each transect for a total of 20 plots. Tree basal area (derived from dbh) and tree density were recorded for all individuals of each species from each plot. Importance values (IV 200) for each species were derived by summing relative basal area and relative density. Density of shrubs and saplings (woody plants < 6 cm dbh and > 50 cm high) was determined from a 0.005-ha circular plot nested at the center of each tree plot. Importance values for shrubs and saplings were calculated by summing relative density and relative frequency.

Frequency and cover data for herbaceous species, seedlings (woody plants < 50 cm), and vines were collected from twelve 0.25-m^2 quadrats in each tree plot. Importance values for each species (IV 200) were calculated by summing relative cover and relative frequency. The ground-cover plots were located along a line transect at 1 m intervals along a randomly determined radius of the tree plot. Only plants rooted in the quadrats were recorded. Sample data collected with similar methods from flatwoods adjacent to the Jackson Slough floodplain forest in a previous study (Taft and others 1995) were used to contrast with the floodplain forest communities. In the flatwoods study, tree plots were independent (not along a transect) and for ground cover 25 quadrats were sampled across a random diameter of each tree plot. Botanical nomenclature follows Mohlenbrock (1986). Classification of natural communities follows White (1978).

Data Analysis

Relationships of diversity and abundance within and among vegetation strata were examined with correlation analysis. Stand compositional and structural stability was examined using a similarity index (Similarity = 2C/A+B, whereas C = species in common, A is species number from subcanopy, and B is species number from canopy) (Mueller-Dombois and Ellenberg 1974).

Vegetation Analysis by Ordination

Ordinations shown for this study are biplots with species and site scores plotted in the diagrams. The ecological meaning of the axes when only species-sample data (and no environmental data) are available are interpretations based on the arrangement of species in the ordinations. The axes can be interpreted as representing gradients of environmental variables and the patterns of species abundance and plot associations in the biplots thus represent the response to the environmental gradients. For some ordinations in this study, to elucidate the relationships among species of floodplain forest and adjacent flatwoods, data previously collected from the flatwoods at Jackson Slough Woods were used (Taft and others 1995).

Vegetation data for each stratum was examined by Detrended Correspondence Analysis (Gauch 1982), using CANOCO ver. 4.0 (ter Braak and Smilauer 1998) in order to characterize the species and plots associations in the floodplain forest and flatwoods habitats. Detrended Correspondence Analysis (DCA) is a multivariate ordination method for indirect gradient analysis based on a unimodal response model and is often used with vegetation data to describe communities, particularly when samples (as in this study) come from across a range of habitats (ter Braak 1995). Gradient lengths on the first DCA axis were greater than 4.9 s.d. for all strata in this study justifying use of a unimodal model rather than a linear-response model such as with Principal Components Analysis. Cluster analysis (PC-ORD ver. 2; McCune and Mefford 1995), a method used to define groups of plots based on similarities, was used for the tree sample data. DCA and cluster analysis were used together to objectively identify plant communities based on tree sample data and how these may relate to natural community classification.

RESULTS AND DISCUSSION

Quantitative Sampling of Floodplain Forest Canopy Stratum

Twenty-six species of woody plants, including trees and a few large grape vines, were identified in the canopy stratum. Based on importance values, *Acer saccharinum* (see Appendix for authority and corresponding common plant names) is the most abundant tree species followed by *A. negundo, Carya laciniosa, Ulmus rubra, Quercus macrocarpa,* and *Fraxinus pennsylvanica* (table 1). Total tree density was 398/ha and total basal area was 31.8 m^2/ha. Stand basal area is slightly less than an old and old second-growth floodplain forest at the northern extent of the coastal plain (34.4 m^2/ha) in southern Illinois (Robertson and others 1978), but greater than southern wet forests (22.6 m^2/ha) at their northern extent in southern Wisconsin (Curtis 1959), and considerably greater than most old growth dry upland forests, flatwoods, or savannas (e.g., Fralish and others 1991, Taft 1997).

Size-class distribution patterns indicate that *A. saccharinum* dominates the intermediate size classes (fig. 1a); however, there are few silver maples in the smaller size classes indicating that presently there is little successful recruitment to the canopy stratum. The pattern for silver maple tree density indicates trees are common in all plots except those nearest and furthest from the river. However, the average diameter of individual trees is greatest near the river. Consequently, while there are fewer trees near the river, their size (and presumably average age) is the greatest in the floodplain; average diameter declines in intermediate zones in the floodplain and increases somewhat away from the river (fig. 1b).

The pattern suggests that establishment and recruitment of silver maple may have changed over time creating a mosaic of scattered old trees near the river, relatively dense stands of smaller (and presumably younger) trees in intermediate zones, and intermediate size and density in the floodplain farthest from the river. As the sample transects ascend a few meters towards the flatwoods community (plot 9 in

Table 1.—*Characteristics of canopy tree species of the floodplain at Jackson Slough Woods in the Kaskaskia River Corridor study area. BA = basal area, IV = importance value, RD = relative density, RBA = relative basal area.*

Species	BA/ha (m-sq)	Density per ha	IV 200 (RD+RBA)
Acer saccharinum	9.269	81	49.48
Acer negundo	1.789	95	29.52
Carya laciniosa	6.201	27	26.27
Ulmus rubra	1.004	46	14.71
Quercus macrocarpa	2.943	9	11.51
Fraxinus pennsylvanica	2.534	11	10.73
Quercus stellata	2.010	12	9.33
Celtis occidentalis	1.124	21	8.81
Carya ovata	0.521	16	5.66
Platanus occidentalis	1.042	7	5.03
Ilex decidua	0.063	15	3.97
Quercus palustris	1.085	2	3.91
Cercis canadensis	0.082	14	3.78
Carya tomentosa	0.446	6	2.91
Morus rubra	0.054	10	2.68
Juglans nigra	0.477	3	2.25
Quercus velutina	0.603	1	2.15
Ulmus americana	0.360	3	1.89
Celtis laevigata	0.083	6	1.77
Vitis sp.	0.016	3	0.80
Populus deltoides	0.063	2	0.70
Sassafras albidum	0.012	2	0.54
Prunus serotina	0.008	2	0.53
Diospyros virginiana	0.005	1	0.27
Vitis aestivalis	0.004	1	0.27
Vitis cinerea	0.004	1	0.27
Quercus rubra	0.004	1	0.26
TOTALS	**31.82**	**398**	**200.00**

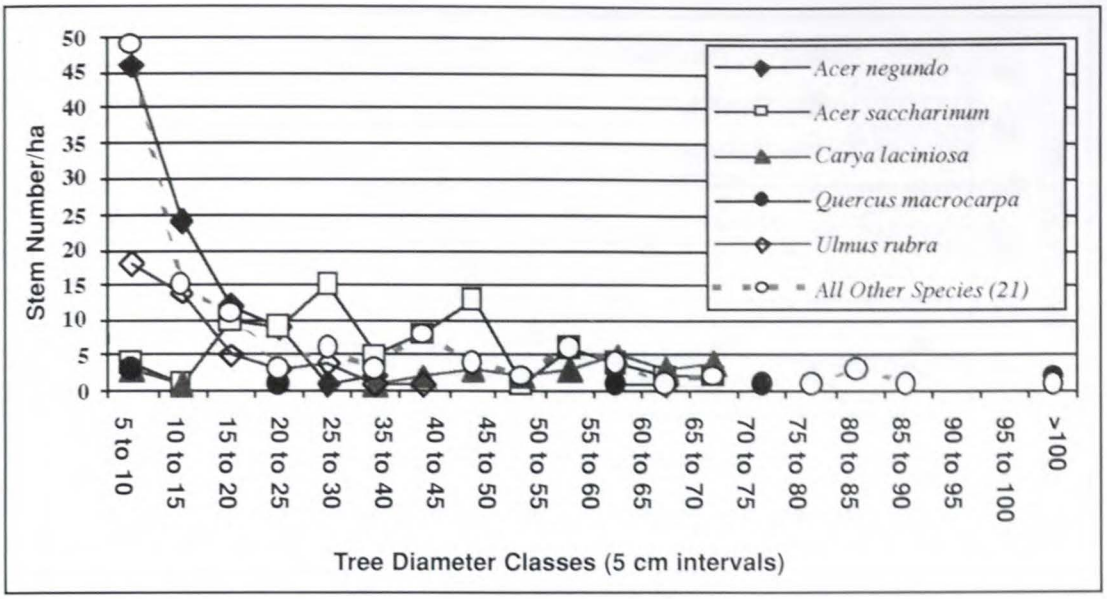

Figure 1a.—*Distribution of tree size classes among dominant floodplain forest tree species at Jackson Slough Woods.*

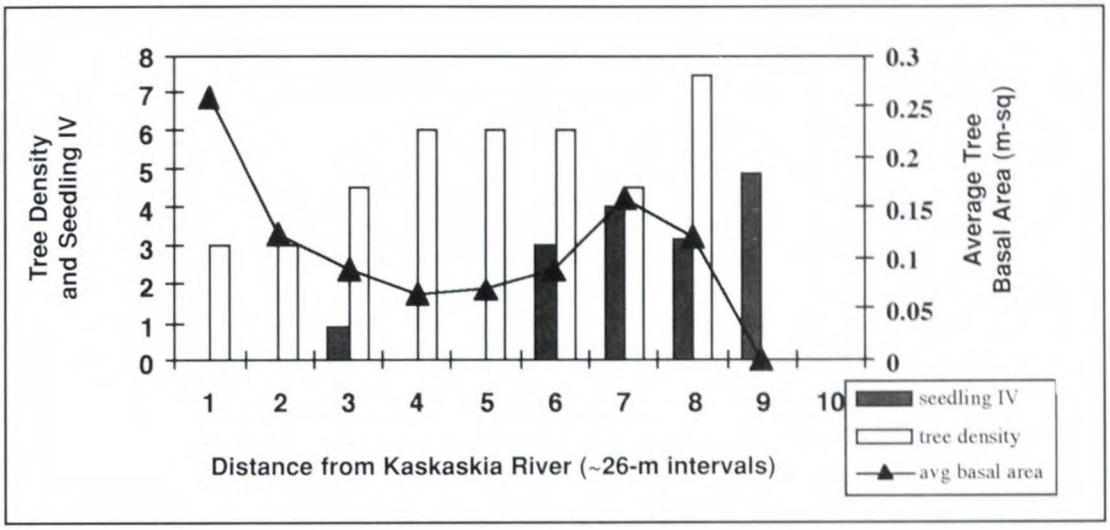

Figure 1b.—*Pattern of tree density, average diameter, and recruitment of silver maple* (Acer saccharinum) *seedling recruitment in old-growth floodplain forest habitat. The distance intervals refer to plot numbers and the data are averaged among two transects. Tree plot size was 0.05-ha; seedling IV refers to the total importance value (sum of relative freqency and percent cover).*

both transects), there are no silver maple trees although seedlings are present. These patterns may reflect variation in flooding season and duration over time due to a meandering river channel. However, hydrological changes as a result of damming the Kaskaskia River upstream at the Carlyle Reservoir may limit flood levels and duration downstream and alter the recruitment patterns for some species such as *A. saccharinum*.

Compositional changes favoring more flood and silt-tolerant tree species like *Acer saccharinum* appear to have occurred since settlement along many Illinois streams (King and Johnson 1977, Nelson and others 1994). Siltation from agricultural lands into the floodplain of the Kaskaskia River and tributaries also may have promoted similar changes in floodplain vegetation within the KRC, at least prior to construction of the Carlyle Reservoir. Floodplain forests in the KRC include some stands, young and mature ages, that are dominated by *A. saccharinum*. However, in contrast to many floodplain forests in glaciated portions of Illinois, oak and hickory species are among the dominant trees in several sites in

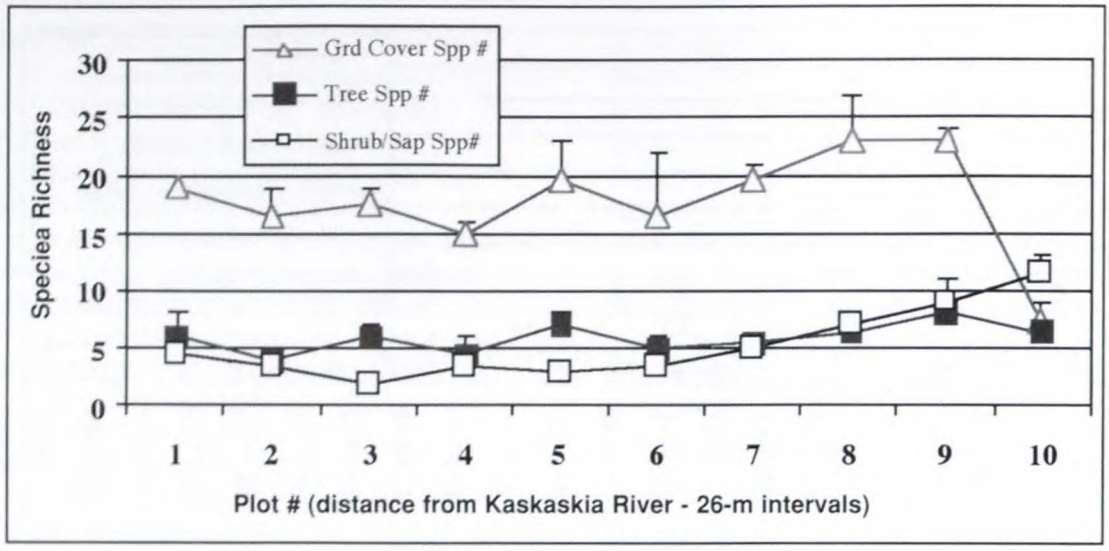

Figure 1c.—*Species richness patterns among vegetation strata along distance gradient from Kaskaskia River to transition with flatwoods upland forest habitat. Values are averages from two parallel transects. Ground-cover species richness was determined from twelve 0.25-m² quadrats per transect at each distance interval (24 plots/interval); shrub/sapling species richness is average from two 0.005-ha circular plots at each interval; tree species richness is average from two 0.05-ha circular plots at each interval. Error bars are standard error.*

the KRC. While *A. saccharinum* is the most dominant species in the sample data, based on total basal area the second and third ranking species are *Carya laciniosa* and *Quercus macrocarpa*, respectively (table 1). Neither species is abundant among individual size classes but both are present as a few trees among small and larger classes suggesting ongoing recruitment that may sustain both species (fig. 1a).

Abundance patterns for *C. laciniosa* and *Q. macrocarpa* show that they are most common in plots where *A. saccharinum* is in low abundance (fig. 2) suggesting these species occupy different habitat niches in the floodplain. At transect 1, *A. saccharinum* is most abundant near the river and becomes less common farther away while *Q. macrocarpa* and *C. laciniosa* become more dominant farther from the river. However, at transect 2, *A. saccharinum* becomes most abundant at points farther from the river while *Q. macrocarpa* and especially *C. laciniosa* are more common near the river. This suggests that the two transects capture some of the spatial variation of the floodplain where local overflow channels and ponding occur at locations distant from the river.

Detrended Correspondence Analysis (DCA) for the canopy stratum reveals clear separation along the first axis (eigenvalue of 0.898) between vegetation in the floodplain forest from the adjacent flatwoods and places plots and tree species found in the transition zone (e.g., *Prunus serotina, Q. velutina, Q. stellata*, and *Carya ovata* in plots 9 and 10 on each transect) in an intermediate position (fig. 3a). Axis II separates plots with species such as *Q. macrocarpa, Q. palustris, Carya laciniosa* from plots characterized by *Platanus occidentalis, Populus deltoides*, and *Acer saccharinum*, species often associated with greater flood frequency and duration. Based on species and plot positions in the ordinations, Axis I is interpreted as a moisture gradient and Axis II is interpreted as a flood-disturbance gradient.

Cluster analysis was used to objectively identify plant species assemblages based on the sample data and how these may relate to natural community classification. With tree data, cluster analysis identified seven groups, based on nearest floristic similarity, and these groups are indicated in the graphical depiction of the DCA (fig. 3a). A key to the species acronyms shown in the ordinations is in the Appendix. From these analyses and the interpretation that the axes in the DCA represent essentially wetness gradients, Group 1 would include plots exposed to the wettest conditions for the greatest duration among all clusters. These plots represent *wet floodplain forest*. Note that in transect 1, plots classified as *wet floodplain forest* (plot 1 and plot 3) are near the river (18 m and 70 m, respectively) while in transect 2, plots classified

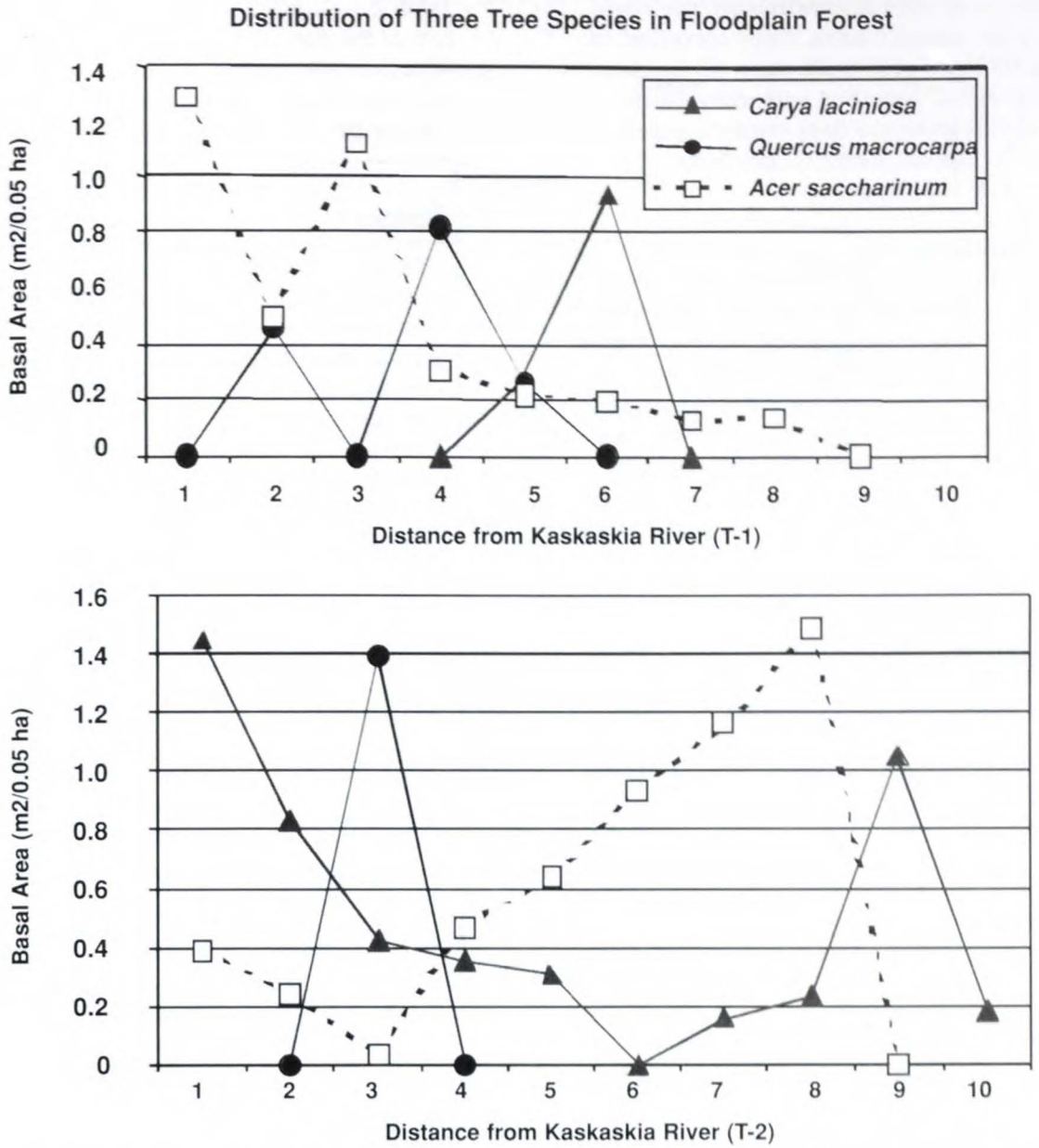

Figure 2.—*Distribution of dominance patterns (basal area m-sq/0.05 ha) for* Acer saccharinum *(Acesai),* Carya laciniosa *(Carlac), and* Quercus macrocarpa *(Quemac), the three most dominant species based on basal area in the floodplain forest samples at Jackson Slough Woods (floodplain). Distance units refer to plot numbers and represent approximately 26-m distance intervals from the Kaskaskia River. T1 refers to Transect 1 and T2 refers to Transect 2.*

as *wet floodplain forest* (plots 6, 7, and 8) occur at locations more distant from the river (147 m to 200 m). Groups 2 and 4 represent *wet-mesic floodplain forest* while Groups 3 and 7 are transitional between the floodplain and adjacent flatwoods (Groups 5 and 6) and can be classified as *mesic floodplain forest*. Richness of tree species across the floodplain and these natural communities remained relatively constant (fig. 1c).

Shrub/Sapling (Subcanopy) Stratum

Twenty-four species of woody plants, including woody vines, shrubs, and saplings, were recorded in the shrub/sapling stratum (table 2). *Toxicodendron radicans* was the most abundant species in the shrub/sapling sample followed by *Campsis radicans, Ulmus rubra, Celtis occidentalis*, and *Ilex decidua*. Stem density was estimated to be 7,440 stems/ha. Excluding the two woody vines, the estimate for saplings and shrubs (mostly saplings) is 3,240 stems/ha.

Only two stems of *Acer saccharinum*, the dominant tree in the sample area, were recorded in the sapling stratum and both were in the same plot. These results, together with data from canopy stratum, indicate that current conditions in the floodplain forest do not favor recruitment of *A. saccharinum*.

Qualitative similarity, a measure of stand stability based on species present both in the subcanopy (i.e., sapling) and canopy stratum (excluding very small species and vines), yielded an index of 72.7 percent. Quantitative similarity, using the least common importance value, yielded an index of 34.47 percent. These values vary only slightly from similarity estimates in the adjacent flatwoods community (78 percent and 21 percent, respectively for qualitative and quantitative indices). Floodplain forest canopy species absent from the subcanopy sample are *Platanus occidentalis, Quercus palustris, Ulmus americana, Populus deltoides*, and *Diospyros virginiana* (a single tree). These trees were all from the wet and wet-mesic floodplain forest zone. Other species absent from the subcanopy include *Juglans nigra, Q. stellata*, and *Q. rubra*, species limited to the transition zone near the flatwoods.

DCA of the shrub/sapling stratum reveals clear separation along the first axis (eigenvalue of 0.851) between vegetation in the flatwoods from floodplain forest (fig. 3b). As with the canopy stratum, based on species and plot positions in the ordination, Axis I is interpreted as a moisture gradient. The species associated with the left side of the first axis (wettest portion of gradient) include *Campsis radicans, Toxicodendron radicans, Fraxinus pennsylvanica, Acer negundo*, and *Ilex decidua*. These are species typically associated with wet and wet-mesic floodplain forest (e.g., plots T1-1, T1-7, T2-6, T2-7, T2-8). Species and plots on the right of Axis I include species of the flatwoods community (e.g., *Prunus serotina, Sassafras albidum, Q. velutina, Q. alba, Carya ovata*, and *Rubus allegheniensis*). Plots and species located intermediately on the first axis were primarily from the mesic floodplain forest transition zone from floodplain forest to flatwoods (e.g., *Morus rubra, Cercis canadensis, Ulmus rubra, Carya tomentosa*) but also included *Carya laciniosa* and *Quercus macrocarpa*. Richness of subcanopy (shrub/sapling) species is low throughout the floodplain but increases in the transition zone (fig. 1c).

Table 2.—*Characteristics of the shrub/sapling stratum of the floodplain forest at Jackson Slough Woods in the Kaskaskia River Corridor study area, St. Clair County, Illinois. IV = importance value, RD = relative density, RF = relative frequency*

Species	Density/ ha	% Frequency	IV 200 (RD+RF)
Toxicodendron radicans	2,480	60	44.76
Campsis radicans	1,720	50	32.64
Ulmus rubra	1,060	60	25.68
Celtis occidentalis	480	60	17.88
Ilex decidua	360	40	12.46
Acer negundo	130	45	10.32
Cornus drummondii	310	25	8.93
Fraxinus pennsylvanica	250	20	7.17
Morus rubra	160	25	6.91
Carya laciniosa	80	30	6.79
Cercis canadensis	70	20	4.75
Celtis laevigata	30	15	3.26
Carya tomentosa	30	10	2.31
Euonymus atropurpureus	30	10	2.31
Quercus macrocarpa	20	10	2.17
Sassafras albidum	70	5	1.89
Prunus serotina	60	5	1.76
Acer saccharinum	20	5	1.22
Quercus bicolor	20	5	1.22
Quercus rubra	20	5	1.22
Carya ovata	10	5	1.09
Carya texana	10	5	1.09
Quercus imbricaria	10	5	1.09
Quercus velutina	10	5	1.09
TOTALS	**7,440**		**200.00**

Ground-Cover Stratum

A total of 81 species of vascular plants were recorded in the ground-cover stratum, including 69 and 52 species for transects 1 and 2, respectively. *Laportea canadensis* was the most abundant species occurring in 40 percent of the quadrats (table 3) with about 14 percent of the total IV. Other species with IV > 5.0 in descending rank order of abundance included *Toxicodendron radicans, Ranunculus septentrionalis, Viola pratincola, Galium aparine, Phlox divaricata, Pilea pumila, Campsis radicans, Carex festucacea*, and *Ulmus rubra* (seedlings). *Carex grisea*, if adding the sterile material believed to be this species (see *Carex* cf. *grisea* in table 3) to the identified material, also would rank among the most important species. *Carex* had the most species among genera in the ground-cover sample with 10. No other genus had more than 3 species (*Galium* and *Vitis*).

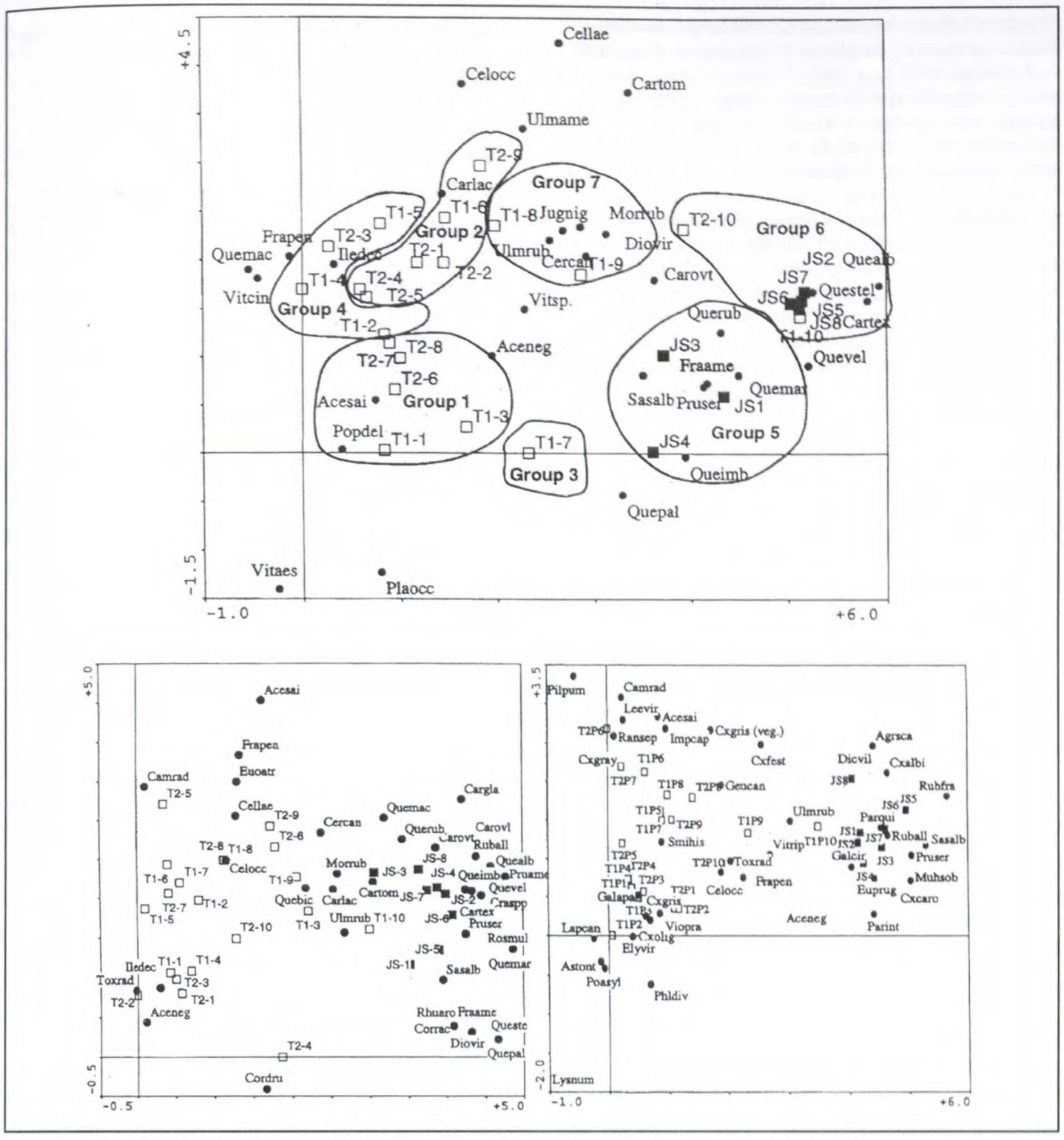

Figure 3a.—*Detrended Correspondence Analysis of canopy stratum of floodplain forest and adjacent flatwoods. Flatwoods plots (JS-1 through JS-8) are indicated by solid squares; plots from floodplain (T1 and T2, plots 1-10) are indicated by open squares. Groupings identified through cluster analysis that are most similar, based on nearest neighbor similarity, also are shown. See Appendix for a key to the species acronym codes.*

Figure 3b.—Detrended Correspondence Analysis of shrub/sapling stratum of floodplain forest and adjacent flatwoods. Flatwoods plots (JS-1 through JS-8) are indicated by solid squares; plots from floodplain (T1 and T2, plots 1-10) are indicated by open squares. See Appendix for a key to the species acronym codes.

Figure 3c.—*Detrended Correspondence Analysis of ground-cover stratum of floodplain forest and adjacent flatwoods. Flatwoods plots (JS-1 through JS-8) are indicated by solid squares; plots from floodplain (T1 and T2, plots 1-10) are indicated by open squares. See Appendix for a key to the species acronym codes.*

Seedlings of *Acer saccharinum* were relatively scarce, occurring in about 7 percent of quadrats and always with less than 5 percent cover (i.e., mostly individual seedlings). These seedlings largely were limited to a zone ranging from about 150 to 230 m from the river (fig. 1b) and were uncorrelated with density of *A. saccharinum* trees ($R^2 = 0.0067$). Occurrence of silver maple seedlings showed a statistically significant inverse correlation with total percent ground cover ($R^2 = 0.22$, df = 19, $p < 0.05$). *Acer saccharinum* is considered to be a shade intolerant species and is noted to be intolerant of competition from other ground cover species (Fowells 1965). Existing abundance patterns among strata suggest an approximately 1 percent survival from one stratum to the next.

Estimates for total seedling number of *A. saccharinum* are about 2,833/ha based on 7 percent frequency of typically single seedlings among 240 quadrats. Sapling density was estimated to be about 20/ha while only 1 and 4 stems/ha were found in the two smallest tree size classes (to 15 cm dbh). With these survival patterns, existing seedling density may be insufficient to maintain *A. saccharinum* at the present abundance in the canopy. These trends may be the result of reduced flood frequency and duration and consequently less available bare soil for successful establishment of *A. saccharinum* seedlings or recruitment of seedlings into the subcanopy or canopy stratum. Regulation of flooding in the watershed may influence long-term distribution patterns of some species among the overstory strata including the current dominant *A. saccharinum*.

Only a single non-native species, *Lysimachia nummularia*, was recorded in the sampling effort, occurring in about 3 percent of the ground cover quadrats. *Lysimachia nummularia*, a perennial herbaceous vine, is a European species occurring in wet floodplain forests throughout Illinois and the Midwest. It is not known to set fruit in Illinois and reproduces entirely through vegetative means.

Among physiognomic groups, perennial forbs dominated with 26 species and about 44 percent of the importance value, followed by woody vines with 8 species and about 19.5 percent of the IV, annual forbs with 8 species (11.5 percent of IV), perennial sedges with 10 species (10.8 percent), tree seedlings with 12 species (7.5 percent), perennial grasses with 5 species (4.2 percent), shrub seedlings with 3 species (0.5 percent), herbaceous vines with 2 species (0.5 percent), biennial forbs with 2 species (0.13 percent), and ferns with 1 species (0.04 percent) among ground cover stratum.

Table 3.—*Summary of quantitative sample data from the ground cover stratum of the floodplain forest at Jackson Sloug Woods. Only species with IV 200 > 1.0 are shown; remainder are available upon request from author. IV = importance value, RF = relative frequency, RC = relative cover.*

Species	% Frequency	% Cover	IV 200 (RF+RC)
Laportea canadensis	40.0	15.95	27.94
Toxicodendron radicans	46.7	11.53	23.83
Ranunculus septentrionalis	52.9	9.43	22.52
Viola pratincola	50.0	4.01	15.22
Galium aparine	37.9	3.96	12.68
Phlox divaricata	17.1	2.88	7.06
Pilea pumila	17.9	2.53	6.81
Campsis radicans	17.5	2.43	6.59
Carex festucacea	7.1	3.70	6.03
Ulmus rubra	15.4	1.86	5.46
Aster ontarionis	10.8	2.04	4.74
Carex grisea	8.8	2.32	4.66
Leersia virginica	8.3	1.96	4.14
Fraxinus pennsylvanica	15.0	0.83	4.10
Smilax hispida	14.2	0.83	3.94
Carex grayii	5.8	2.18	3.89
Carex oligocarpa	6.3	1.30	2.89
Impatiens capensis	9.2	0.37	2.34
Carex cf. grisea	5.4	0.88	2.20
Celtis occidentalis	8.8	0.19	2.03
Poa sylvestris	5.4	0.70	1.98
Lysimachia nummularia	2.9	1.02	1.86
Elymus virginicus	3.3	0.94	1.85
Parthenocissus quinquefolia	5.8	0.46	1.77
Acer saccharinum	7.1	0.15	1.64
Vitis riparia	5.8	0.19	1.44
Aster lateriflorus	3.8	0.50	1.38
Carex tribuloides	1.7	0.82	1.36
Geum canadense	4.2	0.29	1.23
TOTALS		**StDev**	
# Species/0.25 m-sq plot	**4.9**	**1.32**	
Total % cover	**79.9**	**29.40**	
Total species richness	**81.0**		

Species density (average number of species) for the 240 quadrats sampled in the floodplain averaged 4.86 (sd 1.32) per 1/4-m^2 quadrat. Species richness among the 12 quadrats sampled in each tree plot averaged 17.7 (sd 5.16). Cover of ground-layer vegetation averaged 79.9 percent (sd 29.4) among tree plots and 80.8 percent overall. Species density and richness were highly correlated ($R^2 = 0.68$, $p = 0.0001$). Species density and percent cover also showed

significant correlation ($R^2 = 0.26$, $p = 0.02$); however, there was no correlation between species richness and percent cover. Patterns for species richness among canopy, subcanopy, and ground-cover strata (fig. 1c) show ground cover was the most diverse stratum throughout the floodplain. However, species richness of the ground cover declined sharply in the transition zone to flatwoods and was exceeded there by shrub/sapling species richness.

Based on sample data collected in 1989 (Taft and others 1995), ground cover in the adjacent flatwoods had lower diversity and cover compared with the floodplain forest (e.g., mean species density was 2.38, cover was 25 percent). Total ground-cover richness from 200 quadrats in the adjacent flatwoods was 71 compared to 81 species in 240 quadrats in the floodplain forest. In oak-dominated flatwoods and barrens communities there tends to be an inverse relationship between overstory tree density and ground cover diversity (Taft and others 1995, Taft and Solecki, in press), particularly at sites lacking recent fire history. There is no such relationship between over-story structure and patterns of diversity in the ground-cover stratum in the floodplain communities sampled ($R^2 = 0.02$, df = 19).

DCA of the ground-cover stratum reveals, similar to the other strata, clear separation along the first axis (eigenvalue of 0.862) between the ground-cover vegetation in the flatwoods from the floodplain forest (fig. 3c). Species associated with wet to wet-mesic floodplain forests (e.g., *Laportea canadensis, Campsis radicans, Pilea pumila, Ranunculus septentrionalis, Viola pratincola, Aster ontarionis, Poa sylvestris, Galium aparine, Leersia virginica, Impatiens capensis, Carex grisea, C. grayii, C. oligocarpa*) predominate the left side of the ordination so again the first axis is interpreted as representing a moisture gradient (drier to the right side). Plots located intermediately on the first axis include species with fairly broad ecological amplitude (e.g., *Carex festucacea, Toxicodendron radicans, Geum canadense, Vitis riparia, Ulmus rubra*). However, interpretation of the second axis is less clear as species associated with the "wettest" tree plots (Group 1, fig. 3a) are scattered along the length of Axis II (fig. 3c).

CONCLUSIONS

The floodplain forest is comprised of a mosaic of wet and wet-mesic floodplain forest. Mesic floodplain forest occurs in transitional areas with flatwoods. The most salient evidence for change in forest composition is among *Acer saccharinum*, which was found to be the dominant species, but for which there was little evidence of successful reproduction or recruitment. Other species of the wet to wet-mesic floodplain forest canopy that were absent from the subcanopy include *Populus deltoides, Platanus occidentalis, Ulmus americana*, and *Quercus palustris*. A total of 87 species were sampled among canopy, shrub/sapling, and ground cover strata within the floodplain forest. Only a single nonnative species, moneywort (*Lysimachia nummularia*), was recorded in the vegetation sampling effort. The ground cover was the richest stratum with 81 species; perennial forbs dominated the ground cover with 26 species and 44 percent of the importance value. The canopy and shrub/sapling stratum had 26 and 24 species, respectively. There was no relationship between overstory structure and ground cover diversity patterns.

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APPENDIX

List of acronyms used in ordinations with corresponding scientific and common names.

Acronym	Scientific Name	Common Name
Aceneg	*Acer negundo* L.	box elder
Acesai	*Acer saccharinum* L.	silver maple
Agrsca	*Agrostis perennans* (Walt.) Tuckerm.	autumn bent grass
Astont	*Aster ontarionis* Wiegand	Missouri aster
Camrad	*Campsis radicans* (L.) Seem.	trumpet creeper
Cargla	*Carya glabra* (Mill.) Sweet	pignut hickory
Carlac	*Carya laciniosa* (Michx.) Loudon	kingnut hickory
Carovl	*Carya ovalis* (Wangenh.) Sarg.	false shagbark hickory
Carovt	*Carya ovata* (Mill.) K. Koch	shagbark hickory
Cartex	*Carya texana* Buckl.	Texas hickory
Cartom	*Carya tomentosa* (Poir.) Nutt.	mockernut hickory
Cellae	*Celtis laevigata* Willd.	sugarberry
Celocc	*Celtis occidentalis* L.	hackberry
Cercan	*Cercis canadensis* L.	redbud
Cordru	*Cornus drummondii* C. A. Mey	rough-leaved dogwood
Corrac	*Cornus racemosa* Lam.	grey dogwood
Craspp	*Crataegus* sp.	hawthorn
Cxalbi	*Carex artitecta* Mack.	blunt-scaled oak sedge
Cxcaro	*Carex caroliniana* Schwein.	short-scaled green sedge
Cxfest	*Carex festucacea* Schk.	fescue oval sedge
Cxgray	*Carex grayi* Carey	common bur sedge
Cxgris	*Carex grisea* Wahlenb.	wood gray sedge
Cxgris2	*Carex cf. grisea* Wahlenb.	wood gray sedge
Cxolig	*Carex oligocarpa* Schk.	few-fruited gray sedge
Dicvil	*Dichanthelium villosissimum* (Nash) Freckm.	old field panic grass
Diovir	*Diospyros virginiana* L.	persimmon
Elyvir	*Elymus virginicus* L.	Virginia wild rye
Euoatr	*Euonymus atropurpureus* Jacq.	wahoo
Euprug	*Eupatorium rugosum* Houtt.	white snakeroot
Fraame	*Fraxinus americana* L.	white ash
Frapen	*Fraxinus pennsylvanica* Marsh.	green ash
Galapa	*Galium aparine* L.	annual bedstraw
Galcir	*Galium circaezans* Michx.	wild licorice
Geucan	*Geum canadense* Jacq.	white avens
Iledec	*Ilex decidua* Walt.	swamp holly
Impcap	*Impatiens capensis* Meerb.	jewelweed
Jugnig	*Juglans nigra* L.	black walnut
Lapcan	*Laportea canadensis* (L.) Wedd.	Canada wood nettle
Leevir	*Leersia virginica* Willd.	white grass
Lysnum	*Lysimachia nummularia* L.	moneywort
Morrub	*Morus rubra* L.	red mulberry
Muhsob	*Muhlenbergia sobolifera* (Muhl.) Trin.	rock satin grass
Parint	*Parthenium integrifolium* L.	wild quinine
Parqui	*Parthenocissus quinquefolia* (L.) Planch.	Virginia creeper
Phldiv	*Phlox divaricata* L.	blue phlox
Pilpum	*Pilea pumila* (L.) Gray	clearweed
Plaocc	*Platanus occidentalis* L.	sycamore
Poasyl	*Poa sylvestris* Gray	woodland blue grass
Popdel	*Populus deltoides* Marsh.	cottonwood
Pruame	*Prunus americana* Marsh.	American plum
Pruser	*Prunus serotina* Ehrh.	black cherry
Quealb	*Quercus alba* L.	white oak
Quebic	*Quercus bicolor* Willd.	swamp white oak
Queimb	*Quercus imbricaria* Michx.	shingle oak

Acronym	Scientific Name	Common Name
Quemac	*Quercus macrocarpa* Michx.	bur oak
Quemar	*Quercus marilandica* Muenchh.	blackjack oak
Quepal	*Quercus palustris* Muenchh.	pin oak
Querub	*Quercus rubra* L.	red oak
Questel	*Quercus stellata* Wangh.	post oak
Quevel	*Quercus velutina* Lam.	black oak
Ransep	*Ranunculus septentrionalis* Poir.	swamp buttercup
Rhuaro	*Rhus aromatica* Ait.	aromatic sumac
Rosmul	*Rosa multiflora* Thunb.	multiflora rose
Ruball	*Rubus allegheniensis* Porter	black raspberry
Rubfra	*Rubus flagellaris* Willd.	creeping dewberry
Sasalb	*Sassafras albidum* (Nutt.) Nees.	sassafras
Smihis	*Smilax hispida* Muhl.	bristly catbrier
Toxrad	*Toxicodendron radicans* (L.) Kuntze	poison ivy
Ulmame	*Ulmus americana* L.	American elm
Ulmrub	*Ulmus rubra* Muhl.	slippery elm
Viopra	*Viola pratincola* Greene	common blue violet
Vitaes	*Vitis aestivalis* Michx.	summer grape
Vitcin	*Vitis cinerea* Engelm.	winter grape
Vitrip	*Vitis riparia* Michx.	riverbank grape

RIPARIAN FOREST COMMUNITIES OF THE LOWER KASKASKIA RIVER BOTTOMS

Susan P. Romano, James J. Zaczek, Karl W.J. Williard, Sara Baer, Andrew D. Carver, and Jean C. Mangun[1]

ABSTRACT.—Hydrologic alterations due to dam construction may have altered the floodplain ecology of the Lower Kaskaskia River. Seven forest communities within the study site were identified. Floodplain communities include *Acer negundo-Celtis occidentalis-Acer saccharinum, Acer saccharinum-Acer negundo, and Celtis occidentalis-Ulmus americana. Quercus* and *Carya* species were found in both floodplain and upland areas. *Quercus stellata* flats and *Acer saccharum* occupied upland portions of the study site. This community similarity analysis provides a basis for further research on the effects of flood duration, timing, and depth on the current and historical riparian forest community.

Apparent changes in forest vegetation within the Kaskaskia River floodplain present questions about the effects of flood duration and flood depths on the health of the associated riparian forest. A dam was constructed on the Lower Kaskaskia River in 1967 near Carlyle, Illinois for flood control and river navigation. Extensive hydrologic changes are evident when comparing historic and current river gauge data (fig. 1). Landowners and resource managers are concerned about forest composition, tree growth, and regeneration that may be altered by regulated flooding patterns.

Hydrologic alterations within watersheds can be a result of changes in climate, groundwater use, changes in vegetation patterns, and flow diversions. Hydrologic modifications by dams such as high flow, low flow, and flooding duration are distinct for a particular dam, and should not be generalized (Williams and Wolman 1984). Annual maximum discharge for the Carlyle Dam has changed radically from the pre-dam discharge. Rain events produced highly variable water discharge before the dam was constructed, compared to a much less variable and significantly reduced annual maximum discharge following construction (fig. 1).

Altered floodplain hydrology can influence changes in forest communities in several ways. Tolerance to flooding is dependent upon tree species, tree size (seedling, sapling, or mature tree), and period of inundation (Kozlowski 1984). Flood tolerance for many tree species throughout the United States is summarized in Whitlow and Harris (1979). Tree survival may depend on the vigor of the tree before flooding, timing, and depth of flooding. Floodplain forest vegetation changes can occur in response to increased xeric conditions due to the altered hydrologic regime (Walters and others 1980). Silt deposition from floodwaters can increase the elevation of the floodplain and improve soil drainage resulting in dryer conditions (O'Neil and others 1975). Plant succession in some areas may be modified so that earlier plant community types may be replaced by less flood tolerant species (Brinson and others 1981). Seed distribution from flood dispersal mechanisms change with the altered flooding patterns (Middleton 1995), and flood stabilization reduces seedling establishment for some riparian forest species (Fenner and others 1985, Scott and others 1995). Biotic or abiotic, these factors are directly related to the presence, absence, timing, and duration of flooding.

Thirty-four years have passed since the construction of the Carlyle Dam. This provides an opportunity to study the effects of hydrologic alterations on this particular riparian forest community. Under the assumption that certain environmental conditions produce similar forest

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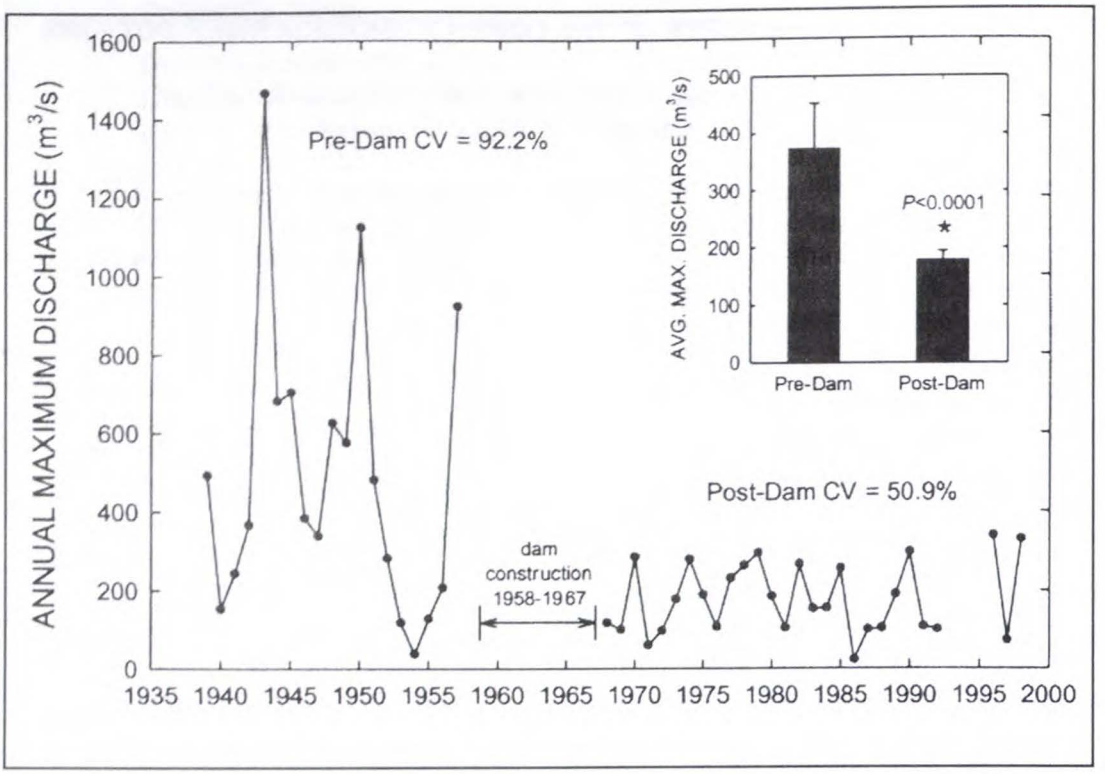

Figure 1.—*Annual maximum discharge from a USGS gauging station on the Kaskaskia River located 94 m downstream of Carlyle Lake Dam.*

communities, cluster analysis was applied as an exploratory analysis technique to detect groupings of similar forest plot data. Conditions that may impact forest communities include elevation, periods of inundation, and timber harvesting. Similarity groupings are identified as a forest community type and described. The objective of this paper is to provide community descriptions of current forest vegetation in the lower Kaskaskia River Basin.

METHODS

Study Site

The Kaskaskia River is a tributary of the Mississippi River; its source in east-central Illinois, flowing south-west and entering the Mississippi River south of St. Louis, Missouri. The Lower Kaskaskia River is the focus of this study, located 40 miles east of St. Louis. This portion of the Kaskaskia is surrounded by the largest contiguous expanse of forest in Illinois (Evans and others 1995). The forests of this area are of the Oak-Hickory Forest region, Northern Division, Prairie Peninsula Section. Within this region are low floodplains dominated by *Acer saccharinum* (silver maple), *Ulmus americana* (American elm), and *Populus deltoides* (cottonwood) (see table 1 for authorities). *Quercus palustris* (pin oak) is found in groupings or as isolated trees within wet areas (Braun 1950).

The study site is located at Venedy, Illinois, along the Kaskaskia River 20 miles south of the Carlyle Dam and about 4 miles south of the Venedy Station, United States Geological Survey (USGS) river gauge station. Elevation of the 80 acre Venedy site ranges from approximately 118.35 to 122.9 m mean sea level. The forest community consists principally of *Acer saccharinum* in the primary floodplain, *Quercus* and *Carya* species in the secondary floodplain, and *Q. stellata* (post oak) in the higher elevations of the study area. This site was selected because of its proximity to a gauge station with current and historical data, range in elevation, and a cooperative landowner.

Plots were randomly selected by applying the RAND function in EXCEL to the length and width of the sample area to obtain plot coordinates. If plots overlapped in the field, a random number and direction were chosen to offset the location. Plots were measured between July 10, 2001 and August 20, 2001 within the 80-acre study site. Forty circular overstory plots, each 0.04 hectares, were marked with steel fence posts. Plot locations were identified using a Global Positioning System. Tree species were

Table 1.—*Tree density and basal area for harvested and non-harvested areas of the Lower Kaskaskia River near Venedy, Illinois, USA*

Species	Non-harvested area		Harvested area	
	Tree density trees/ha	Basal area m²/ha	Tree density trees/ha	Basal area m²/ha
Acer negundo L.	1400	57.57	2400	77.01
Acer saccharum Marsh.	375	12.69		
Acer saccharinum L.	1825	145.00	925	47.09
Betula nigra L.	25	1.48		
Carya illinoiensis (Wangenh.) K. Koch	100	7.49	25	4.34
Carya laciniosa (Michx. f.) Loud.	25	5.31	25	6.20
Carya ovata (Mill.) K. Koch	800	37.66	225	7.07
Carya texana Buckl.	50	1.74		
Carya tomentosa (Nutt.)	425	32.95	75	8.32
Celtis occidentalis L.	1575	70.24	1150	57.07
Cercis canadensis L.	25	0.19	75	0.76
Cornus florida L.	25	0.23		
Crataegus sp.	25	0.26		
Diospyrus virginiana L.	50	1.58	50	2.88
Fraxinus americana L.	400	37.90	100	18.23
Fraxinus pennsylvanica Marsh.	150	16.70	150	5.71
Gleditsia triacanthos L.	50	7.35	75	0.78
Maclura pomifera (Raf.) Schneid.			25	0.41
Platanus occidentalis L.			25	0.38
Prunus serotina Ehrh.	125	1.36		
Quercus alba L.	425	23.83	50	7.28
Quercus bicolor Willd.	300	25.16	25	0.53
Quercus imbricaria Michx.	100	11.46		
Quercus macrocarpa Michx.	25	2.07	75	3.80
Quercus palustris Muenchh.	225	34.57	25	0.44
Quercus rubra L.	550	55.80		
Quercus stellata Wangenh.	750	91.71		
Quercus velutina Lam.	375	43.54		
Salix nigra Marsh.	25	2.67		
Ulmus americana L.	375	12.69	700	21.34
Ulmus rubra Muhl.	50	3.91	125	2.42

identified according to Mohlenbrock (1986). Diameter at breast height (dbh) was measured to the nearest 0.1 cm for all trees greater than 9 cm. Latitude and longitude determined through the GPS were combined with the USGS Digital Elevation Model (DEM) to provide plot elevation data.

Data Analysis

COMPAH (combinatorial, polythetic, agglomerative, hierarchial) was implemented to separate community groups based on basal area and species. COMPAH is a cluster analysis program that groups samples into dominance types (Boesch 1977, Fralish and others 1993). COMPAH first determines the resemblance, or similarity, of each of the plots sampled. The Bray-Curtis similarity coefficient, used in this analysis, is one of the most extensively used similarity coefficients in ecology (Boesch 1977, Gaugh 1982). Group averaging was also applied to the data, and is a common clustering method used in ecology. This method is "the mean of all resemblances between members of one group to members of another" (Boesch 1977). Group averaging further distinguishes the plots based on the patterns of resemblance, and determines the community groups (Boesch 1977).

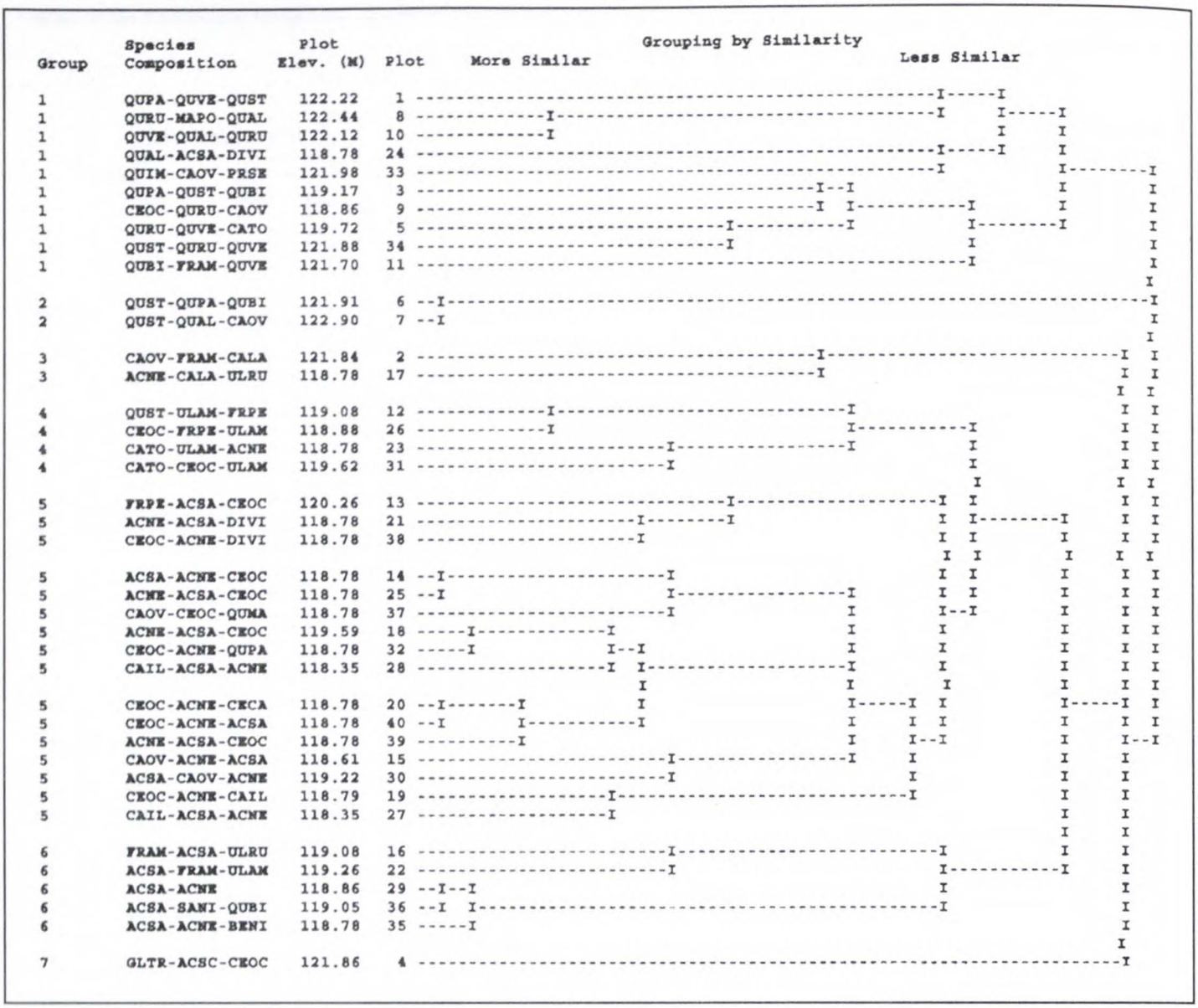

Figure 2.—*Forest community groups generated from COMPAH analysis of species and basal area for each plot within the watershed of the Lower Kaskaskia River near Venedy, Illinois. Species are indicated by genus-species code based on first two letters, respectively.*

RESULTS

COMPAH grouped overstory tree species and basal area data into 7 forest communities based on similarity (fig. 2). Groups 5, 6, and 4 contain primarily floodplain forest species. Group 1 spans both floodplain and hilltop, dominated by *Quercus* species. Group 3 also contained upland and floodplain species, but with *Carya laciniosa* in common, and Groups 2 and 7 are upland forest communities.

The primary floodplain forest community was *Acer negundo-Celtis occidentalis-Acer saccharinum* (Group 5). Within this grouping, *Acer negundo* occurred often (14/16 plots), and at higher basal area relative to other species. *Celtis occidentalis* occurred (14/16 plots) similar to *A. negundo*, but generally at lower basal area. *Acer saccharinum* also occurred very frequently (13/16 plots), also at generally lower basal area than *A. negundo*. This forest community occupies the broad, primary floodplain across the lower elevations ranging from 118.35-120.26 m.

Group 6 is another floodplain community. Basal area of *A. saccharinum* was greater in this group than Group 5, and *A. negundo* generally was present at a lower basal area in Group 6 than in Group 5. Both *Acer* species occurred in all five plots in the grouping. *Acer negundo* occurred at basal areas ranging from 0.5 to 3.9 m^2/ha, with high densities ranging from 150 to

600 stems/ha. The span of elevation for this group was 118.78-119.26 m.

Group 4 characterized by the presence of *Celtis occidentalis* and *Ulmus americana*, found in all plots. This group is distinctive in that it lacks the *A. saccharinum* component found in Groups 5 and 6. The elevation span for this group is 118.78-119.62 m. Group 1 is dominated by a variety of *Quercus* species, and spans an elevation from 118.78-122.44 m. *Quercus bicolor*, *Q. alba*, and *Q. rubra* were oak species of the low and moderate elevation plots, while *Q. velutina*, *Q. alba*, and *Q. stellata* are present at higher elevations.

The strictly upland forest community groups were Group 2 and Group 7. The remaining Group 3, represented a small sampling of *Carya* sp., and may not be related to elevation. Group 2 is a *Quercus stellata* community, and Group 7 as one plot containing *Acer saccharum* Marsh. Group 3, was neither upland nor lowland, with a *Carya laciniosa* influence.

Forest overstory species of the above communities have specific flood tolerances, indicated by their presence at various elevations in this floodplain. Figure 3 illustrates the distribution of individual species along the elevation gradient, based on presence-absence data. Soft maples were specific to low lying areas: *Acer negundo* was limited to a range of 118.35–119.59 m, and *Acer saccharinum*, 118.35–120.26 m. Some species were broadly dispersed across a range of floodplain elevations and into upland elevations. *Fraxinus pennsylvanica*, *Ulmus rubra*, *Celtis occidentalis*, and *U. americana* were generally present at the lower elevations and were often a component within the predominately *A. saccharinum* and *A. negundo* elevations, but were not limited to the lower elevations.

Oak and hickory dominated higher elevations with *Quercus bicolor*, *Carya laciniosa*, *C. tomentosa*, *Fraxinus americana*, *Quercus palustris*, *Q. alba*, *Q. rubra* L., and *Q. stellata* (fig. 3). A narrow elevation distribution was apparent for the flood intolerant *Q. velutina*, distributed at 119.72–122.44 m. *C. ovata* was the most ubiquitous species, present at elevations that supported soft maples, and also present in the high elevation *Q. stellata* community.

Forest management is another factor that may influence species composition. Table 1 separates data from an area harvested about 20 years ago from the data collected in an area that was not harvested. The area harvested was primary floodplain, located adjacent to the Kaskaskia River. Similar floodplain species were present in the harvested area, with the inclusion of *Platanus occidentalis* and *Maclura pomifera*, and the exclusion of *Betula nigra*.

DISCUSSION

Flooding patterns and the tolerance of species to flooding influence the establishment and survival of forest communities. Floodplain plant community structure is a result of the effects of flooding over time, set back to an earlier successional state by extreme floods, and accelerated when flooding is reduced (Junk and others 1989). Factors other than flooding are also involved. Robertson and others (1978) indicated that soil characteristics and topographic form are also very important factors in floodplain forest community patterns. Silt deposition and seed dispersal mechanisms also influence the composition of floodplain forest communities (O'Neil and others 1975, Walters and others 1980, Middleton 1995).

Flood tolerance of species may explain some of the similarity within community groupings. The largest primary floodplain community of the study site is composed of flood tolerant species. The *Acer saccharinum-Acer negundo-Celtis occidentalis* community group was common across the primary floodplain of the study area. Flood tolerance may have a large influence on these composition patterns. Mature *A. saccharinum* can survive flood conditions for long periods, up to 149 days in one study (Bell and Johnson 1974), and for 3 years in another study (Yeager 1949). *Acer negundo* was also tolerant to 149 days of flooding, and *Celtis occidentalis* survived 109 days of flooding (Bell and Johnson 1974).

The *Acer saccharinum-Acer negundo* community (Group 6) is an interesting similarity grouping. Silver maple is a very flood tolerant species, and large diameter trees more tolerant of flooding than trees of smaller diameter. This may result in an even-age floodplain forest structure of large diameter *A. saccharinum* (Oldenburg 1980). The lack of diversity in these plots could also be an artifact of successional state, current flood patterns, or soil characteristics.

Ulmus americana has a wider range of tolerance for conditions found in the floodplain than any other floodplain forest species (Hosner and Minkler 1960). It is considered to be somewhat

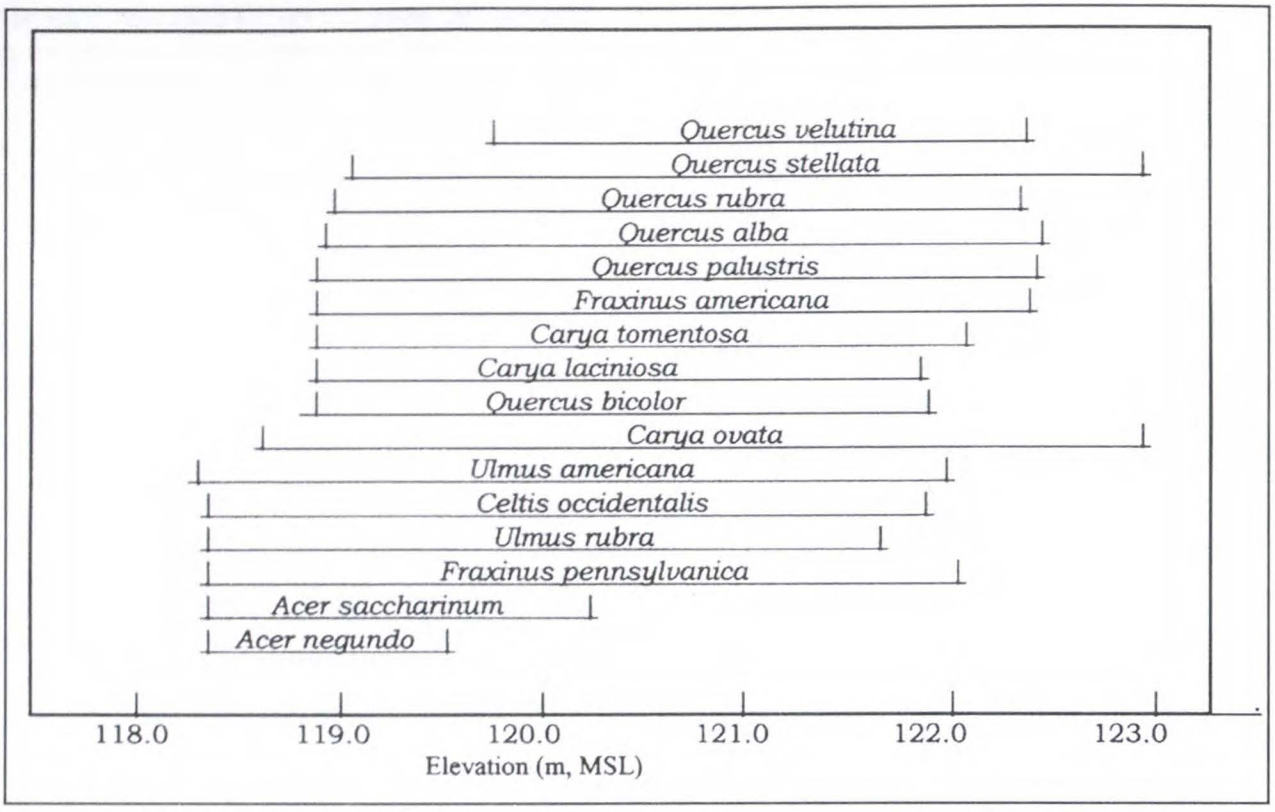

Figure 3.—*Species distribution across the elevation gradient present in the watershed of the Lower Kaskaskia River near Venedy, Illinois.*

flood tolerant by Bell and Johnson (1974). This unique combination of a wider tolerance range, and slightly less flood tolerance relative to other floodplain species may contribute to the presence of *U. americana* in the *Celtis occidentalis-Ulmus americana* community.

The *Carya* sp. group is a small similarity grouping of only two plots. Hickory was found across all but the lowest floodplain areas. Further data collection at other sites will determine if this grouping, although quite different from other community groups, is prevalent among the floodplain forest of this site.

Upland forest communities contained species that were intolerant to moderately tolerant of flooding. The *Q. stellata* flat is seasonally wet and dry, on a perched water table (Taft and others 1995). Only one plot contained *A. saccharum*, an upland species.

Quercus species were present on the slopes above the floodplain and throughout the floodplain below. More tolerant *Quercus* species were found in the floodplain. The *Quercus* similarity grouping contained all *Quercus* species across the range of plot elevations. This indicates a dispersal of oak without a particular community or elevation relationship. Resource managers are concerned about the perceived lack of oak regeneration in the floodplain. Our results indicate that several species of oak are still present as overstory trees in the floodplain. These include *Q. stellata*, *Q. rubra*, *Q. alba*, *Q. palustris*, and *Q. bicolor*. Midstory and seedling strata will be analyzed for regeneration of oak in the floodplain, and at higher elevations within the site. Further investigations will determine if the floodplain forest is succeeding to an *A. saccharinum* forest community without an oak component in the lower strata.

Harvesting may be another factor to consider. The similarity analysis (fig. 2) indicated that the harvested floodplain plots within the study site were similar to the unharvested areas. Harvested plots were numbers 24, 26, 21, 25, 28, 40, 39, 27, and 22. The summary of species, basal area, and density in table 1 indicated that although there were differences in basal area and density, the same species were present in the harvested area as were in the old growth floodplain, with only a few exceptions. This similarity indicates that species tolerance to flooding may have a greater impact than harvesting in determining species composition of riparian forest communities, making succession concepts questionable.

CONCLUSION

This study provides community descriptions that demonstrate some relationships among elevation, hydrology, and forest tree species in this Illinois bottomland forest. Further research will incorporate data from two additional Lower Kaskaskia River floodplain sites. This data will include species and basal area of the overstory and mid-story strata, soil factors, and inundation periods. Specifically, *Quercus* species correlations to these factors will be investigated. Historical growth and inundation relationships will be determined through dendrochronology techniques. This community analysis is the first step in our efforts to define the past, present, and future community patterns within the modified floodplain of the Lower Kaskaskia River.

ACKNOWLEDGMENTS

Our sincere thanks to Glen Schuetz, landowner, for his cooperation and assistance with this research, and Mark Koch, District Forester, and Marty Kemper, Wildlife Biologist, Illinois Department of Natural Resources, for their help in locating potential study sites. Our appreciation is extended to Dr. Philip Robertson for his assistance in interpreting the analysis, and to the anonymous manuscript reviewers. This research was made possible by a grant through the Illinois Department of Natural Resources, Conservation 2000 Program.

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LATE-SEASON FLOODING REGIMES INFLUENCE THE ACCRETION AND PARTITIONING OF NITROGEN AND BIOMASS IN SILVER MAPLE SEEDLINGS

Christopher M. Kaelke and Jeffrey O. Dawson[1]

ABSTRACT.—To better understand how late-season flooding affects survival, growth, N accretion, and N partitioning in silver maple, we subjected 1-year-old seedlings to continuous (July-November), summer (July-September), and fall (September-November) root-flooding regimes outdoors in plastic tanks. Only root growth was affected by flooding as root biomass measured in September and November was reduced in the summer and continuously flooded seedlings. Seedling N accretion was decreased only in the continuous flooding treatment. Fall flooding resulted in earlier resorption of foliar N and leaf abscission than in controls and recovering summer-flooded seedlings, thereby insuring conservation of foliar N under fall flood stress conditions. Our results indicate that silver maple seedlings can tolerate summer and fall flooding and reveal how silver maple seedlings grow, take up, and partition N as they adapt to flooding under altered hydrologic regimes.

Silver maple (*Acer saccharinum* L.) is the dominant bottomland hardwood species throughout much of the central Midwest (Bell 1974, Peterson and Bazzaz 1984, Dunn and Stearns 1987, Yin and others 1997, Nelson and Sparks 1998, Yin 1998). It commonly occurs in lowland areas in major river floodplains and at the edges of streams and lakes (Gabriel 1990). Silver maple not only is the dominant canopy tree species, but also makes up the largest proportion of the seedling and sapling classes in many areas, particularly in the upper Mississippi River floodplain north of St. Louis, Missouri (Yin and others 1997, Yin 1998).

Silver maple's ecological importance has greatly increased as natural hydrological regimes have been altered for river navigation, flood protection, and reservoir construction (Yin and others 1997, Nelson and Sparks 1998, Sparks and others 1998, Yin 1998). These alterations have resulted in flooding regimes that are irregular in seasonality, duration, and intensity. They also have led to an increase in the average annual low-water level, predisposing more bottomland areas to prolonged soil saturation at any time of the year (Yin and others 1997, Sparks and others 1998). At the same time that silver maple has increased in importance under the altered flooding regimes, woody species diversity in bottomland hardwood forests has declined in the region (Yin and others 1997, Nelson and Sparks 1998).

Silver maple's capacity to tolerate various flooding regimes probably is an important factor responsible for its increasing dominance (Bell 1974, 1980) under conditions in which it can be flooded during any month of the year. Yet no studies have examined the ability of silver maple to continue to grow and assimilate nitrogen when flooding occurs late into the growing season. Nor have the effects of late- season flooding regimes on the partitioning of biomass and nitrogen within maple seedlings been investigated. Seedling physiology is important in itself, without correlation to mature plant function, because of the need to understand the basis for seedling establishment, survival, and growth, all necessary prerequi-sites for recruitment of silver maple into larger size classes.

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A few controlled, greenhouse studies demonstrated that silver maple seedlings were able to survive either soil saturation or complete submergence for periods of 3 weeks to 2 months (Hosner 1960, Hosner and Boyce 1962, Loucks and Keen 1973). Peterson and Rolfe (1983) found that maple seedlings survived up to 40 days of complete submersion, but that biomass production, especially in the roots, was reduced more by a June flood than by an August flood. Peterson and Bazzaz (1984) reported that complete submersion was more injurious to growth than root-flooding of silver maple seedlings. Soil waterlogging for 2 months resulted in significantly reduced biomass production, particularly in the roots. Flooding of root systems for 1 month also reduced seedling photosynthetic rates, but the degree of reduction did not differ between a mid-May or a late August flood (Peterson and Bazzaz 1984).

In silver maple seedlings root-flooded for 2 months, the total shoot N content was twice that of controls (Hosner and Leaf 1962). However, root N content, initial and final seedling dry weights on a whole-plant basis, and the redox potential of the flooded system were not reported. Pezeshki and others (1999) contended that the degree of soil reduction in the flooded system must be taken into consideration in any study of flooding effects on woody plant nutrient uptake and allocation. Moreover, nitrogen is the most limiting nutrient in many wetland ecosystems (Ponnamperuma 1984). DeLaune and others (1998) stressed that information is needed on the dynamics of N uptake by wetland tree species to better predict their growth and survival under the hypoxic soil conditions imposed by flooding.

In order to better understand how growth, N accretion, and the partitioning of biomass and N in silver maple seedlings are affected by flooding when it occurs late in the growing season, we subjected 1-year-old seedlings of silver maple to summer and fall root-flooding regimes outdoors in tanks containing a nutrient solution. The summer flooding treatment began in early July and continued until early September. The fall flooding treatment was initiated in early September and ended in early November after a killing frost and when daily maximum air temperatures were less than the minima generally required for photosynthesis (Kozlowski and Pallardy 1997). A continuous flood treatment also was initiated in early July and continued through the end of the growing season in order to determine how a longer period of flood stress would impact silver maple's capacity to physiologically function. An unflooded set of maples was maintained as a control.

We hypothesized that relative to controls, flooding would affect the accretion and partitioning of biomass and nitrogen differently as the season progressed, but that silver maple would be able to survive all flooding treatments. To construct seasonal N budgets for silver maple, we assumed that all seedling N accretion was due to N uptake and that the influence of atmospheric N deposition was negligible [National Atmospheric Deposition Program (NRSP-3)/National Trends Network 2001]. The results of our study support the notion that silver maple is tolerant of flooding when it occurs late in the growing season and that the relatively minor effect that flooding had on its carbon and nitrogen economies, except when exposed to prolonged inundation, is a probable reason for its success under altered hydrologic regimes in bottomland ecosystems.

MATERIALS AND METHODS

One-year-old seedlings of silver maple from an Illinois seed source were obtained from a commercial nursery in April, 1999, and potted in 11.3-L plastic pots containing a 1:1:1 (v/v/v) mixture of vermiculite: calcined clay: sand. After a 1 month establishment period in a greenhouse, maple seedlings were transferred outdoors at the Plant Sciences Laboratory on the University of Illinois campus. Seedlings were then grown outdoors for 5 weeks before initiation of flooding treatments on July 1. Maples were arranged in a randomized block design with two blocks and were randomly assigned to one of four flooding treatments:

1) summer: maples flooded above the root collar with nutrient solution (see below) in 500-L plastic tanks from July 1 until September 5, and then allowed to recover;
2) fall: maples flooded from September 5 until November 10;
3) continuous: seedlings flooded from July 1 through November 10; and
4) unflooded control seedlings which were arranged randomly and adjacent to the flooded tanks and placed on saucers.

Within each block, maples in all treatments were randomly re-arranged every 2 weeks to minimize positional effects.

Each month the plastic tanks were drained and immediately refilled with one-quarter strength modified low-N Hoagland's solution (Hoagland and Arnon 1950) with nitrogen provided at a concentration of 14 ppm, similar to that

reported previously for soil in a central Illinois silver maple stand (Peterson and Rolfe 1985). Compounds providing the macronutrients were supplied at the following concentrations (mg/L): K_2SO_4 - 109; $MgSO_4$ - 60; $Ca(H_2PO_4)_2.2H_2O$ - 63; $CaSO_4.2H_2O$ – 86; NH_4Cl - 47. N was supplied as ammonium-N only because it is the form of N that is available in flooded systems due to denitrification of nitrate (Whitlow and Harris 1979, Day 1987). Micronutrients were supplied at one-half strength (Hoagland and Arnon 1950). Unflooded control seedlings were watered daily with distilled water and leached with the same low-N Hoagland's solution every 2 weeks. There was a higher frequency of nutrient solution change for the unflooded maples. This was warranted by the lower total nutrient amount supplied to each unflooded plant in an 11.3-L pot relative to those in the 500-L flooded tanks.

Weekly measurements of solution pH were made for the tanks. At the time of each monthly nutrient solution replacement, the initial solution pH averaged 6.2 ± 0.1 and the pH stabilized at 7.1 ± 0.2 within 1 to 2 weeks after recharging. The redox potential (Eh) of soil in flooded pots was measured periodically with a platinum electrode (Orion Research Inc., Beverly, MA, USA) to verify that a hypoxic system had been established (i.e., Eh ≤ 350 mV, the redox potential at which molecular oxygen is negligible). The Eh of flooded pots averaged 288 ± 32 mV following the addition of 244 mV as an adjustment to a standard hydrogen electrode.

An initial whole-plant harvest of 15 seedlings was conducted on July 1. Subsequent harvests were made on August 4, September 5, October 7, and November 10, for four seedlings (two per block) per treatment. All harvested plants were sorted into leaves, stems, and roots, dried to constant mass at 80° C, and weighed. Tissue samples were ground to 40-mesh in a Wiley Mill (Thomas Scientific, Swedesboro, NJ, USA) and approximately 50 mg of tissue from each sample was acid digested (Cataldo and others 1974). All plant nitrogen was reduced to ammonium-N and total Kjeldahl nitrogen was subsequently measured as ammonium-N colorimetrically on a spectrophotometer at 625 nm (Cataldo and others 1974).

The location of the study site on the campus of the University of Illinois at Urbana-Champaign (40° 07' N, 88° 12' W) has a temperate continental climate typified by cold, dry winters and warm, moist summers with 104 cm of precipitation and 185 frost-free days per year (Illinois State Water Survey, Champaign, Illinois). The mean summer (June - August) daily high temperature is 28° C with a mean daily low of 17° C. Mean winter (December - February) temperatures range from a daily high of 2° C to a daily low of –6° C. The total precipitation for the period of July through the end of October 1999 was 33.0 cm, slightly drier than the 30-year average of 38.3 cm for the same period. Daily maximum and minimum temperatures were used to determine monthly averages for air, the nutrient solution in a plastic tank, and for the soil in an unflooded pot at the study location (table 1).

Means and variances of dependent variables were determined for all treatments at each harvest. A common mean for each dependent variable was reported for the initial, pre-flood harvest in July. Preliminary analyses revealed no differences between blocks and this source of variation was removed from the analysis, yielding four replicates for each treatment at each harvest. All data analyses were conducted with Systat, version 9 (Systat, SPSS Science, Chicago, IL). Biomass and nitrogen content data were natural log transformed and analyzed by one-way analysis of variance separately at each harvest with flooding treatment as the independent variable. Nitrogen ratios (organ N content:seedling N content) were arcsine

Table 1.—*Mean daily maximum and minimum temperatures (°C) for each month from July through October at the study site. Nutrient solution temperatures were measured in a 500-L plastic tank and soil temperatures were taken in a pot containing an unflooded control seedling.*

	Air		Nutrient solution		Soil	
Month	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum
July	33.4	20.8	32.1	23.7	39.8	21.7
August	32.7	17.3	29.8	20.6	35.7	17.6
September	26.2	12.9	24.4	16.0	30.8	13.6
October	19.4	7.9	14.6	10.0	17.3	7.9

transformed. If the overall F test within a particular harvest was significant ($P \leq 0.05$), a Fisher's protected LSD was used to compare treatment means. At a given harvest, treatment means were reported as significantly different in the results when $P \leq 0.05$.

RESULTS

Survival and Flood-Induced Morphological Characteristics

Maple seedlings in every flooding treatment had 100 percent survival through the end of the growing season. All non-harvested seedlings resumed growth the following spring. Maples flooded during the summer period displayed hypertrophied lenticels on submerged portions of the stems. However, only two summer-flooded seedlings developed any noticeable adventitious roots on the stem. No fall-flooded maples developed hypertrophied lenticels. Root systems in all flooded treatments had substantial necrosis of both coarse and fine roots, but newly produced adventitious flood roots were visible on all flooded root systems. These roots were white and generally had little branching.

Leaves of summer-flooded maples turned red during the first 3 weeks of flooding, most likely due to anthocyanin production. Upon recovery in the fall, the same leaves on these seedlings became green again. The leaves of fall-flooded maples became red during the first 2 weeks of fall flooding and dropped by October 20. In the continuous flood treatment, leaves turned red during the first 3 weeks of summer flooding, began to abscise during September, and were completely abscised by October 20. Summer-flooded seedlings and unflooded controls had lost all their leaves by the end of the first week in November.

Seedling Growth and Biomass Partitioning

Flooding had little effect on seedling growth throughout most of the study. Mean whole-seedling biomass for each flooding regime did not differ significantly from unflooded controls except at the November harvest (fig. 1). In November, the mean seedling biomass of maples that had been continuously flooded since July was significantly less than the means for all other treatments. Seedling biomass of summer-flooded maples was significantly less than that of controls in November. The mean biomass of fall-flooded plants was not significantly different from the unflooded control means at any harvest.

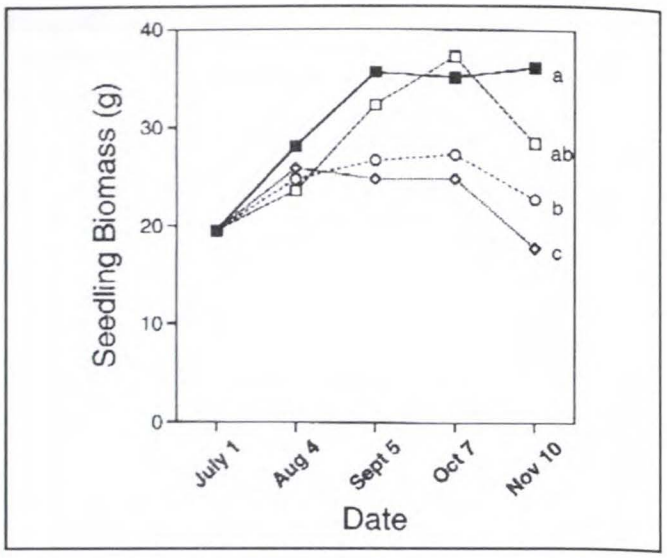

Figure 1.—*Mean whole-seedling biomass (g dry weight) for treatments at each harvest. Treatment means are denoted by black boxes for controls, diamonds for continuous flooding, circles for summer flooding, and open boxes for fall flooding (N=4 for all treatment means). An initial harvest of 15 seedlings was conducted on July 1. The starting dates for the summer and fall flood periods were July 1 and September 5, respectively. Treatment means within an individual harvest not sharing common letters were significantly different ($P \leq 0.05$).*

Flooding did not affect the partitioning of biomass among leaves, stems, and roots throughout most of the study. Neither leaf nor stem biomass of any flooded treatment differed significantly from that of unflooded controls at any harvest (figs. 2a, 2b). At the September and November harvests, mean root biomass of the summer and continuously flooded maples was significantly less than that of controls. The continuously flooded seedlings did not display a net increase in root mass during the fall period (fig. 2c). Root biomass of maples in the fall flooding treatment did not differ significantly from that of controls. In unflooded controls, the biomass of woody tissues increased by 102 percent during the summer period and by 24 percent in the fall.

Nitrogen Accretion and Partitioning

A significant effect of flooding on whole-seedling N content was not evident until the November harvest. Mean seedling N content of the continuously flooded treatment was significantly less than that of controls (fig. 3). At the October harvest, mean leaf N content of both the continuous and fall flooding treatments were significantly lower than those of the controls

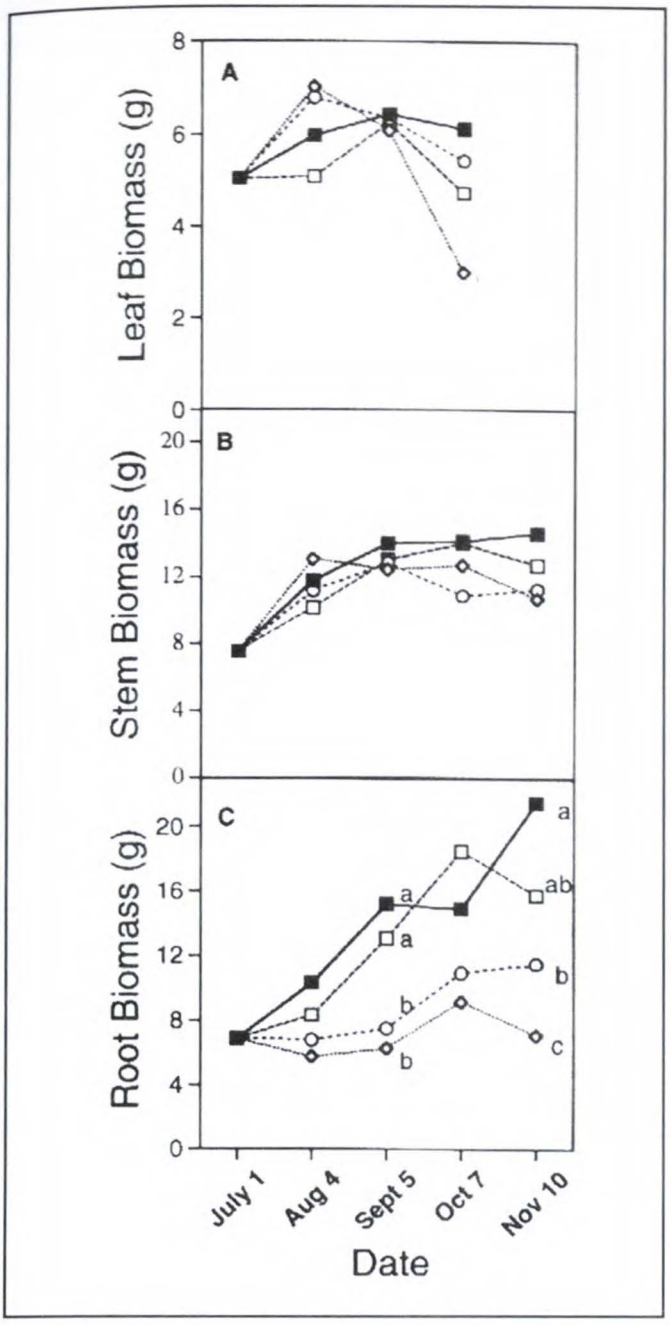

Figure 2.—*A. Mean leaf, B. stem, and C. root biomass (g dry weight) for treatments at each harvest. See figure 1 caption for explanation of symbols. Leaf masses are not included for the November harvest due to complete leaf abscission for all treatments prior to that harvest.*

and summer flooding treatment (fig. 4a). Significant differences in stem N content occurred only for the October harvest. Mean stem N content of the fall-flooded treatment was significantly greater than those of the unflooded controls and the summer flood treatment (fig. 4b). The stem N content of the fall-flooded maples also exhibited a steep upward trend during the first month of fall flooding. Mean

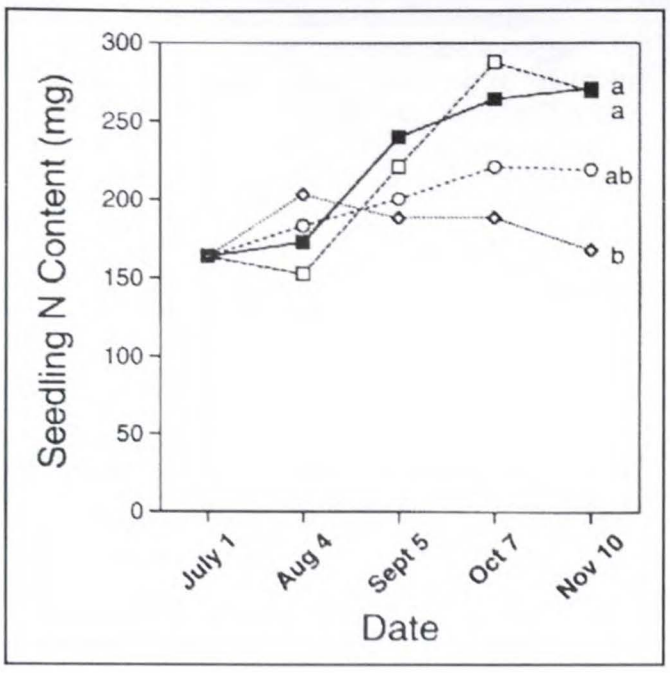

Figure 3.—*Mean whole-seedling N content (mg N) for treatments at each harvest. See figure 1 caption for explanation of symbols.*

root N content was not significantly reduced by flooding until the November harvest (fig. 4c).

Net fall N accretion in woody tissues was substantial among all treatments except for the continuously flooded maples. Woody tissue N content of controls increased by 87 percent in the fall. In the summer and fall-flooded seedlings, woody tissue N content increased by 60 and 95 percent, respectively, during the fall period. The ratio of N partitioned to leaves versus woody tissues was affected by flooding early in the fall flooding period. The proportion of seedling N in leaves (leaf N content:seedling N content) decreased considerably during the first month of the fall period in the continuous and fall-flooded treatments (fig. 5a). At the October harvest, the proportion of seedling N in leaves was significantly less for the continuous and fall flood treatments relative to that of the unflooded controls and summer-flooded seedlings. During the first month of the fall flood period, the proportion of seedling N in woody tissues (woody tissue N content:seedling N content) increased sharply in the continuous and fall flooded treatments (fig. 5b). In October, the proportion of seedling N in woody tissues was significantly greater for the continuous and fall flooding treatments relative to the summer-flooded maples and unflooded controls.

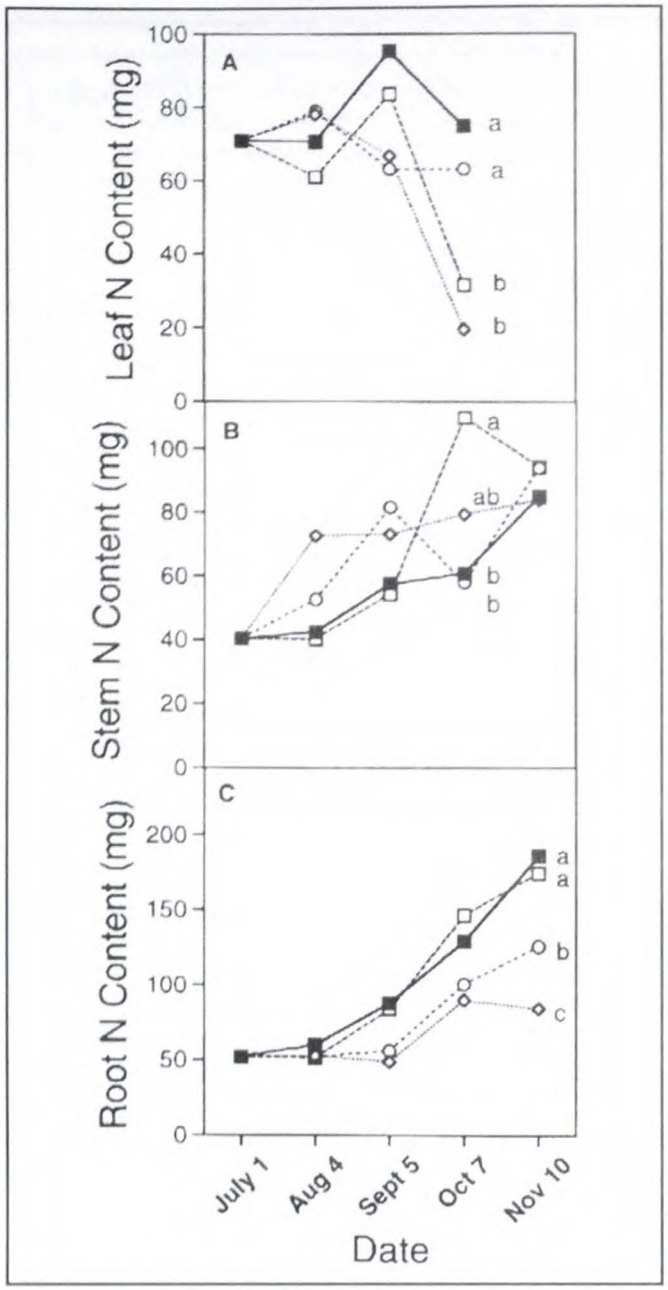

Figure 4.—*A. Mean leaf, B. stem, and C. root N contents (mg N) for treatments at each harvest. See figure 1 caption for explanation of symbols. Leaf N contents are not included for the November harvest due to complete leaf abscission for all treatments prior to that harvest.*

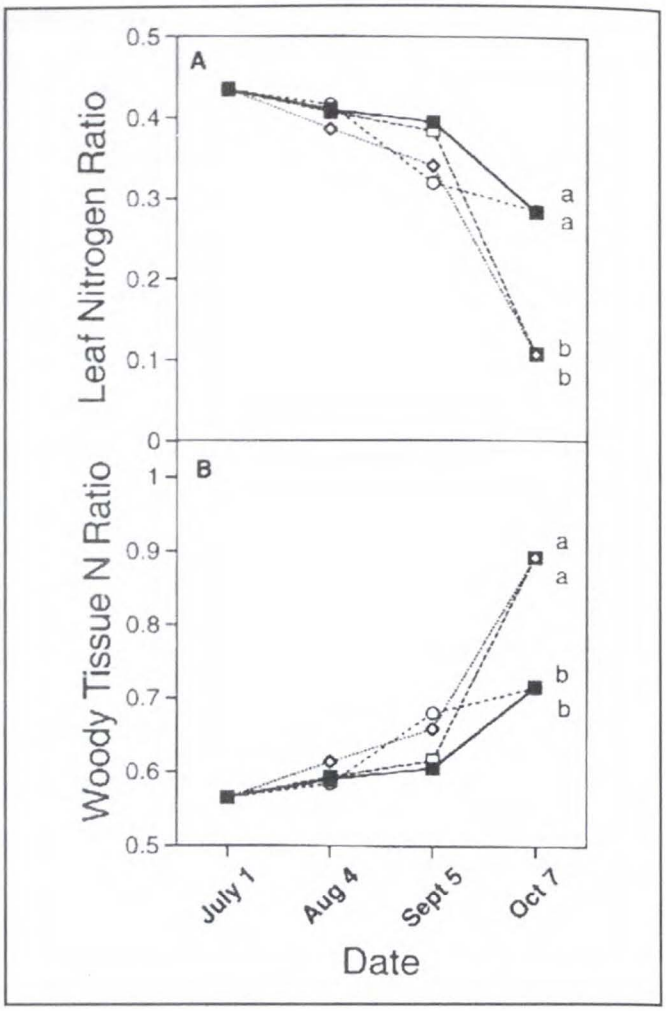

Figure 5.—*A. Mean leaf nitrogen ratio (mg leaf N content:mg seedling N content) and B. mean woody tissue nitrogen ratio (mg woody tissue N content:mg seedling N content) for treatments at each harvest. See figure 1 caption for explanation of symbols. Treatment means within an individual harvest not sharing common letters were significantly different ($P \leq 0.05$). Nitrogen ratios are not included for the November harvest due to complete leaf abscission for all treatments prior to that harvest.*

DISCUSSION

Survival and Flood-Induced Morphological Characteristics

At the outset of this study, we hypothesized that 1-year-old silver maple seedlings would be able to survive summer and fall flooding, but that accretion and partitioning of biomass and nitrogen would differ according to flooding regime. In partial support of our hypothesis, 1-year-old silver maple seedlings were able to completely survive not only summer and fall root-flooding regimes, but also continuous flooding from July through November. Our results for survival are in accordance with previous reports of no mortality for silver maple seedlings exposed to root-flooding or complete submersion for periods of 3 weeks to 2 months (Hosner 1960, Hosner and Boyce 1962, Loucks and Keen 1973, Peterson and Rolfe 1983, Peterson and Bazzaz 1984). The flood-induced hypertrophy of lenticels that we observed in the summer and continuous flooding treatments has been noted by others for silver maple (Hosner and Boyce 1962, Peterson and Rolfe 1983).

However, the lack of development of hypertrophied lenticels in our fall flooding treatment is noteworthy in that it suggests that, when flooded in early fall, silver maple may begin senescence processes prematurely and may not allocate resources to structures that would promote late-season growth. The development of hypertrophied lenticels has been documented as a mechanism by which flood-tolerant species transport atmospheric oxygen to their roots, enabling aerobic root metabolism to continue during flooding (Hook 1984, Kozlowski 1997). This internal transport of atmospheric oxygen may also lead to oxidation of the rhizosphere, thereby decreasing potentially toxic compounds that may develop under flooded conditions.

The flood-stress-induced early abscission of leaves is further evidence that fall flooding may lead to a premature initiation of normal fall senescence processes in silver maple. Leaf abscission in the continuous flooding treatment began in September while that of the fall flooding treatment started in early October. Leaves of plants in both these treatments were completely abscised approximately 3 weeks earlier than those of controls and summer-flooded maples, precluding carbon gain via photosynthesis. In addition, the leaves of the continuously, summer, and fall-flooded maples all turned red during the first 3 weeks of their respective flood treatments, most likely due to chlorophyll loss and anthocyanin production.

Rapid anthocyanin production in leaves may reflect an osmotic adjustment to avoid water stress (Chalker-Scott 1999), but most of the literature on this topic indicates that flood-tolerant woody plants generally do not develop reduced leaf water potentials when flooded (Kozlowski 1997). Anthocyanin production occurs when excess sugars accumulate due to slowing of sugar export via phloem. Flooding of roots can result in stomatal closure (Kozlowski 1997) and reduced phloem transport. However, anthocyanins produced in leaves under stress may also protect the photosynthetic apparatus against light-induced photooxidation (Chalker-Scott 1999). In addition to the seedlings in our study, flood-induced anthocyanin production has been noted in silver maple seedlings in floodplain ecosystems (C.M. Kaelke and J.O. Dawson, personal observations). The adaptive significance of this flood stress response in silver maple warrants further investigation.

Seedling Growth and Biomass Partitioning

We did not expect that late-season flooding would so minimally reduce biomass production. This finding may derive from the fact that much of the current year's new biomass had most likely been accreted before the outset of the study in July (Peterson and Rolfe 1983). Hosner and Leaf (1962) reported that after 62 days of soil saturation, shoot biomass of flooded silver maple seedlings was actually 32 percent greater than controls, but they did not report root biomass. In our study, seedling biomass after leaf drop was lowest in the continuous flooding treatment. This result supports the contention of Peterson and Bazzaz (1984) that flood duration is a major factor affecting physiological function in silver maple.

Yet, also in support of our hypothesis, our study suggested a seasonal factor that may influence silver maple biomass production. In November, the seedling biomass of summer-flooded plants was also significantly reduced relative to unflooded controls. Thus, flooding during the summer did result in a growth reduction that became apparent at the end of the growing season. The pattern of significance for root biomass among the treatments in November was the same as for whole-seedling biomass. Because leaves were completely abscised by the November harvest and stem biomass did not differ among the treatments, this indicates that flood-induced reductions in whole-seedling biomass by the end of the growing season were largely due to reductions in root growth. A decrease in root growth was not apparent in the fall-flooded seedlings. Both Peterson and Rolfe (1983) and Peterson and Bazzaz (1984) found the greatest biomass reduction in silver maple seedlings to be in the flooded roots. In our study, 4 months of continuous root-flooding resulted in no net increase of root biomass during the fall period, most likely due to sloughing of dead root tissues and depletion of nonstructural carbohydrate reserves.

Nitrogen Accretion and Partitioning

An effect of late-season flooding on seedling N content was not evident until November. In the continuously flooded treatment, both whole-seedling and root N content in November were reduced relative to controls. Because treatment differences did not exist for stems in November and leaves had already abscised, the flood-induced reduction in whole-seedling N content was due to reduced root N content. The significantly lower root N content was largely

attributable to the lack of increase in root biomass in the continuously flooded maples during the fall. Therefore, at the end of the growing season, only seedlings that had been flooded continuously for 4 months had reduced seedling N content relative to controls, again underscoring the tolerance of silver maple seedlings to all but the most prolonged flooding regime. Hosner and Leaf (1962) found that shoot N contents of silver maple seedlings grown in saturated soil for 60 days were twice that of controls. Although they did not report root N contents or soil redox potential of their system, their results also are suggestive of silver maple's ability to continue N uptake under the reduced conditions of flooded soils.

Even though the summer and fall-flooded seedlings did not exhibit significantly lower seedling N contents than controls in November, the processes by which they arrived at similar N contents probably differed. During the first month of the fall flooding period, seedlings in both the continuous and fall-flooded treatments exhibited a precipitous decline in leaf N content. In the fall-flooded maples, this decline occurred without appreciable leaf drop whereas in the continuous flooding treatment, the decline was associated with leaf drop. Maples in the summer flooding treatment and the unflooded controls did not experience such a drop in leaf N content until the final month of the study.

At the same time that leaf N content was rapidly declining in the continuous and fall flooding treatments, stem N content increased in the fall-flooded maples. Because there was no leaf drop during the first month of fall flooding when much of the rapid decrease in leaf N occurred (fig. 4a), the foliar N decline in the fall-flooded seedlings was apparently due to flood-induced early resorption of foliar N prior to leaf abscission. The concurrent rapid decrease in leaf nitrogen ratio with a rapid increase in woody tissue N ratio during the first month of fall flooding (figs. 5a, 5b) indicates the occurrence of fall flood-induced foliar N resorption at a rate of efficiency of 62 percent [(September leaf N content - October leaf N content)/ September leaf N content]. Thus, fall flooding may induce early N resorption that could promote resumption of growth the following spring (Kozlowski and Pallardy 1997). The expedited resorption of N insures seedling N conservation with flood-induced early leaf senescence. These results support our hypothesis that N accretion and partitioning would be affected differently by flooding as the growing season progressed.

The potential N resorption pool (September foliar N) as a proportion of total net fall N accretion in woody tissue was similar among treatments (i.e., 76, 77, and 64 percent in the controls, summer-flooded, and fall-flooded maples, respectively). Although both environmental and genetic factors can determine resorption efficiencies, genetic factors seem to be more important for temperate deciduous trees (Vogel and Dawson 1993). Estimated resorption efficiencies for temperate deciduous trees range from 25 to 66 percent (Kozlowski and Pallardy 1997). Based on the high proportional value of N in the potential resorption pool to total N accretion in woody tissue in the fall and the likely range of resorption efficiencies, we conclude that N conservation is substantial in both flooded and unflooded maples.

CONCLUSION AND IMPLICATIONS

Our study demonstrated that silver maple seedlings survived, grew, accumulated N, and therefore, were tolerant of flooding in both summer and fall, but especially in the fall. Fall flooding expedited normal fall senescence processes including resorption of foliar N. Continuous flooding for the final 4 months of the growing season apparently hinders silver maple's capacity to set aside carbohydrate and mineral nutrient reserves to resume vigorous growth at the start of the following growing season. The results of our study suggest that under altered hydrologic regimes, silver maple could maintain seedling establishment, survival, growth, and N accretion better than many sympatric tree species rated as less flood-tolerant (Gill 1970, Whitlow and Harris 1979).

If the management goal is for greater woody species diversity in the bottomlands, then carefully planned measures, such as flood control and creation of floodplain mounds with dredge spoil, to promote growth of species less flood tolerant would be necessary. On the other hand, silver maple has shown promise as a rapidly growing hardwood species suitable for biomass fuel production on marginal lands (Geyer 1989, Walter and Dwyer 2003). Silver maple's versatility in surviving flooding in bottomlands may render it a desirable species for plantings intended to reduce flood damage in riparian corridors (Walter and Dwyer 2003). The capacity to continue accretion rates similar to unflooded seedlings and to favorably alter partitioning of biomass and N during various root-flooding regimes probably are important reasons for silver maple's increasing dominance in midwestern floodplain forests.

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SURVIVAL RESULTS OF A BIOMASS PLANTING IN THE MISSOURI RIVER FLOODPLAIN

W.D. 'Dusty' Walter and John P. Dwyer[1]

ABSTRACT.—A factor essential to successful tree planting in unprotected floodplain environments is survival. Two-year survival results from tree planting in an unprotected floodplain adjacent to the Missouri River are presented. Species planted included silver maple, locally collected cottonwood, and a superior cottonwood selection from Westvaco Corporation. Two spacings, 4 x 4 feet and 8 x 8 feet, were established to evaluate mortality for each species. As a result of recurrent flooding, mortality was high for all tree species. At the end of both the first and second year silver maple had the best survival, averaging 85 and 27.5 percent, respectively. Lessons learned take into account the need to understand the dynamics of floodplain properties prior to reclamation and establishment. These dynamics include the relationship of changes in elevation that result not only in soil depth changes, but also duration of submersion during flood events. Intensive site preparation, as well as documenting the history of flood events, should precede the establishment of tree crops in floodplain environments.

The location of the study site was chosen because it represents the potential of agroforestry in an active floodplain that is subject to periodic flooding. Much of the Missouri River floodplain is in fact not subject to flooding because it is on the protected side of the U.S. Corps of Engineers and privately owned and maintained levee systems. The research site was subject to inundation during flood events. Therefore, it provided an excellent demonstration of how the integration of trees with traditional agronomic cropping practices could help reclaim and protect potentially productive farm ground.

Floodplains are areas adjacent to major waterways that are subject to periodic flooding. These areas have traditionally provided some of the most fertile and productive farmland throughout the United States and are therefore often subject to clearing. The floods that occurred throughout the Midwestern United States during 1993 were of historic (500-year) proportions (Hanback 1994). Because of the duration, velocity, and extent of floodwater, levee failures resulted in extensive damage to agricultural land, water quality, wetland ecosystems, rural and urban infrastructures, and human resources (Scott 1993). In Missouri alone crop losses exceeded $235 million on approximately 1.2 million acres (Center for National Food and Agriculture Policy 1994).

Past research has shown that floodplain corridors with woody trees will slow floodwater velocities, dissipate energy, reduce scouring potential and increase soil shear strengths (Scheifele 1928, Shields and Gray 1993, Young 1994). A study of primary levees along the Missouri River following the 1993 flood showed that, as the width of the woody corridor between the river bank and primary levee increased, the damage both in numbers and size of levee failures decreased (Dwyer and others 1997). Therefore, if trees could be reintroduced to farm landscapes prone to flooding, many benefits can be realized. Trees protect the land and its soil from the erosional effects of water. Benefits from roots include increases in soil stability and increased water infiltration rates (Brooks and others 1991). Leaves, twigs, branches, and trunks, serve to intercept rainfall and slow surface flows prior to the rains' impact on the soil. One mechanism for the reintroduction of trees into traditional agriculture settings is termed agroforestry.

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Agroforestry is defined as "an intensive land-management system that optimizes the benefits from the biological interactions created when trees and/or shrubs are deliberately combined with crops (Garrett and others 1994, 2000)." Agroforestry provides the means of incorporating intensively managed trees and crops on the same land for mutual benefits while at the same time diversifying income opportunities. The purpose of this study was to investigate the potential of Short Rotation Woody Crops (SRWC) as the forestry component in a floodplain agroforestry practice. SRWC, also called short-rotation intensive culture, is a descriptive term for the intensive planting, at close spacings, of fast growing trees to be harvested on short-rotations of 2 to 10 years (Perlack and Geyer 1987, Ranney and others 1987, Colletti and others 1991). Because the harvesting time frame and establishment practices are more similar to agronomic systems than traditional forestry, SRWC is a viable enterprise for incorporation into an agroforestry practice with all the associated benefits. This paper focuses on establishment and survival of trees planted in an active floodplain.

METHODOLOGY

Site Location and Description

The Missouri Department of Conservation (MDC) granted permission to use 20 acres of floodplain ground along the Missouri River in Eagle Bluffs Conservation Area located in Boone County, 6 miles southwest of Columbia near McBaine, Missouri. The study site occupies approximately 18 acres of the piece of land formerly known as the Holiman tract. The location is on the unprotected side of the main levee.

The natural levees, formed from years of sand and silt deposited adjacent to river channels, were broken by the flooding that occurred in 1993. Movement of water and debris in the floodwaters of 1993 and 1995 eroded gaps in the natural levee and resulted in sand splays onto portions of the study site. Those same levee gaps have since allowed annual flood waters (often several times per year) to enter onto the Holiman tract, now an active floodplain. This alluvial floodplain is lower in elevation than the natural levee gap and thus water backs onto the site and ponds.

Deposition affecting this study is predominantly a result of backswamp and overbank flooding. Waters which either overflow river banks or back fill onto lower lying areas (backswamps), resulting in little or no water movement, create situations where overbank deposits form (Schmudde 1963). Thus, periodic flooding creates multiple layered soils of varied depth and differing texture according to duration and movement of floodwaters. Often abrupt soil variation makes soil predictions within small floodplains difficult (Daniels and Hammer 1992).

Soils Information

Ten-baseline soil samples were taken from two transects by extracting soil to a depth of 60 inches. From these samples of the soil horizons, their structure and texture were identified, as well as the depth of the perched water table. Except for the northeastern corner where slope can be closer to 2 percent, the site has a slope of 1 percent or less. In the absence of water current, the only erosion that results is in the form of wind-wave action. Consequently, these soils are mostly depositional material. In general, the soil profile is as follows:

1) AP horizon: 0 to 7 inches, 10YR 3/2, silt loam, 18 percent clay;
2) Bg1: 7 to 24 inches, 2.5Y 3/2, silty clay loam, 32 percent clay;
3) Bg2: 24 to 38 inches, 2.5Y 4/2, silty clay loam, 30 percent clay;
4) Bg3: 38 to 48 inches, 2.5Y 4/2, silty clay loam, 34 percent clay;
5) 2C1: 48 to 54 inches, 2.5Y 5/3, sandy clay loam, 28 percent clay; and
6) 2C2: 54 to 60 inches, 2.5Y 6/3, very fine sandy loam, 19 percent clay.

Across the site the depth of perched water table varied from 34 to 46 inches beneath the surface. This may represent water perched from above or below. Redoximorphic features, indicated in part by the presence of oxidized iron, typically identified these areas.

Study Design

A Complete Randomized Block Design (CRBD) was chosen for tree planting design. This allows for the comparison of three tree species planted at two spacing treatments. The tree species chosen were local cottonwood (*Populus deltoides* Bartr.), a superior cottonwood clone donated by Westvaco Corporation, and silver maple (*Acer saccharinum* L.). Seedlings of these species were each planted at two spacings of 4 x 4 feet and 8 x 8 feet. Each treatment plot measured 96 x 72 feet. These plots, representing each species at one of the two spacings, were randomly placed in larger blocks of 96 x 504 feet (figs. 1 and 2). These larger blocks were replicated four times. Each replication contains a control where the naturally occurring regeneration was recorded.

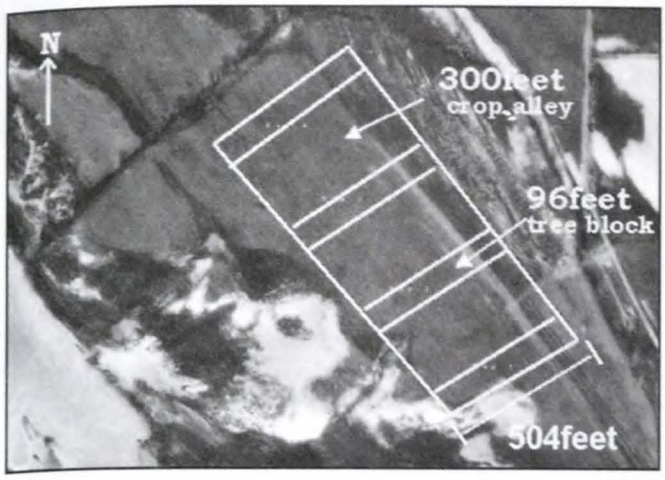

Figure 1.—*Site layout, with tree replication and alley crop areas identified.*

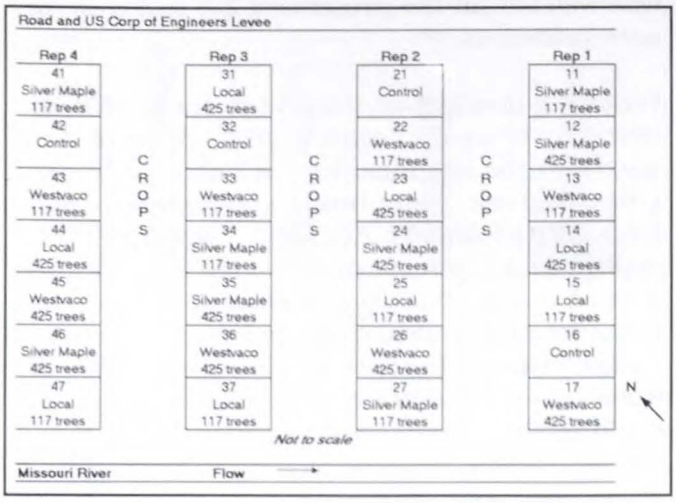

Figure 2.—*CRBD layout of each replication by species and spacing treatment location.*

The total acreage of the four replications that were planted in trees was 4.4 acres. Between each replication was a 300 x 504 feet alley (fig. 1). Within this alley the crop component of the agroforestry system was planted. The species chosen were switchgrass (*Panicum virgatum* L.), alsike clover (*Trifolium hybridum* L.), and redtop grass (*Agrostis stolonifera* L.).

Cottonwood and silver maple were chosen because they are both indigenous species of the floodplain that also exhibit excellent growth characteristics desirable for biomass and other wood products. The total sample size of 2,168 bareroot seedlings and cuttings were comprised of the following species: cuttings of a superior cottonwood clone (Westvaco), cuttings of cottonwood collected adjacent to the site (local), and silver maple bareroot seedlings. Numerous studies have been done to compare various spacings with densities as low as 1,000 trees per acre and some as high as 43,500 trees per acre, the effect on yield versus economic viability (Elk and others 1983, Ranney and others 1987, Torreano and Frederick 1988, Geyer 1989). For comparison, the spacing treatments of 4 x 4 feet and 8 x 8 feet were selected. These two spacing treatments resulted in 425 and 117 trees in each plot, respectively. Tighter spacings dictate earlier harvest or, at minimum, thinning operations. Data on survival was measured by species/spacing treatment.

Treatment Establishment

In preparation for spring planting we did fall site preparation to control undesirable vegetation. This vegetation included willow, cottonwood, and vegetation such as cocklebur, sunflower, smartweed, ragweed, morning glory, nutsedge, panicum grass, and johnson grass. Because of the thickness of existing vegetation we had initially planned to clear the site by controlled burn, but the Missouri Department of Conservation had reservations about burning the area and this method of clearing was not allowed. Our second option, to brush hog and disc the site, was done the fall of 1997 and winter (February) 1998. This resulted in good organic matter incorporation into the soil and initial site preparation was considered complete.

Studies have shown that the control of vegetative competition is essential for maximization of biomass production in SRWC systems (Ranney and others 1987, Torreano and Frederick 1988, Mitchell 1995, Buhler and others 1998). During the establishment phase, Mitchell (1995) points out that survival and growth of unrooted cuttings will be affected by any competition.

Following the planting of silver maple (May 5-12, 1998) Scepter® (registered trademark of American Cyanamid Company) was applied to all tree planting treatments. Scepter is primarily recognized as a pre-emergent herbicide, yet it will control young weeds as a post-emergent if a surfactant is used. Since some trees had broken dormancy and begun to grow, Scepter was applied without surfactant. Using a backpack sprayer and a stovepipe section to cover seedlings, Scepter was sprayed in a 2- to 3-foot radius circle around each of the silver maple seedlings. The cottonwood received a broadcast spray application of Scepter between each row. Though an effort was made to avoid emerging leaves, Scepter has been approved for application overtop of actively growing *Populus* species. The application rate was 3.2 oz. per acre, and

was well within the prescribed 2.8 to 5.6 oz. per acre guidelines.

Following flooding, at the end of June 1998, a new crop of weeds began to grow. Some of the weeds on the site included cocklebur, Johnson grass, dogbane, water hemp, and ragweed. Thus, from July 13 through August 4, Roundup® was applied as a 2 percent mixture. The Roundup was applied using a backpack sprayer and stovepipe to insure no chemical contacted the seedlings and rooted cuttings. Because of the height of some of the cottonwood (over 3 feet tall) the stovepipe could not be used to insure against herbicide contact, so that cottonwood did not receive this second herbicide application.

In addition to herbicide, we also tried to establish a cover crop in order to control undesirable weed growth. At the end of February 1999, we used broadcast spreaders to frost-seed alsike clover into all tree replications. Seeded at a rate of approximately 6 pounds per acre, we hoped to utilize the clover's nitrogen fixing capabilities for benefits to the trees while creating a cover crop which would out compete weeds yet be no taller than 3 feet. At this height the cover crop should not over top the trees.

Data Analysis

Survival of all live trees was recorded for each species and treatment. The presence of dead or not yet fallen leaves, and stem flexibility and color were used as indicators of a potential live tree. To test whether replication, species, or spacing influenced survival a Chi-square test was used on a random sample of 30 trees from each species/spacing treatment. Statistical analysis was performed using SAS (SAS Institute, Inc., Cary, NC) with all significance tests performed at a confidence level of 95 percent, or P=0.05. The row-by-column Chi-square test on survival was used to test the following hypothesis:

1) H_o: Number of live trees does not vary by replicate, species, or spacing; and
2) H_a: Number of live trees does vary by replicate, species, or spacing.

RESULTS

At the end of the first and second growing seasons a complete survey was done of all trees growing within the control treatments. The only control treatment in which natural regeneration occurred was in replication one. By the end of the first growing season, three cottonwoods had become established. Their diameters for the first year were 0.44, 0.56, and 0.42 inches. The same trees had heights of 5.17, 6.25, and 6.0 feet, respectively. By the end of year 2 only two of these trees had survived. Each measured 8.0 feet in height and had diameters of 1.43 and 1.49 inches.

Analysis of Survival

I found no published research to document seedling survival of plantings in active floodplains. Other studies have shown that survival will vary by site. From uplands to riparian zones survival typically falls in the range of 70 to 100 percent for both cottonwood and silver maple (Geyer 1989, Schultz and others 1995, DeBell and others 1996, Geyer and Walawender 1997). In addition to the key role survival plays in the creation of a demonstration site, it is also one of the keys to maximizing production of biomass on a given area, and the other is fast growth. At the end of both the first and second growing seasons survival was measured.

The total number of trees (all species combined) that survived in each replication was low. Of the total 180-tree sample (3 species x 2 spacings x 30 trees sampled per treatment) first year survival from replications 1, 2, 3, and 4 was 32, 28, 32, and 37 percent, respectively (fig. 3). The Chi-square test did not indicate significant differences (P=0.05) in survival between replications.

By the end of the second year, survival percentages by replication had decreased. Figure 3 shows the decrease in survival for each replication, and the tests of second year survival by replication were significantly different. Replication 2, having the lowest percentage survival (2 percent), and replication 4 with the highest percent survival (almost 23 percent) were significantly different. Replications 1 and 3 were statistically similar having survival of 15 and 12 percent, respectively.

When survival within each replication was tested by individual species group, there were significant differences. Across all replications first year survival was significantly higher for silver maple (fig. 4). Comparisons between replications of silver maple showed replication 1, with 97 percent survival, to be significantly higher than replications 2, 3, and 4. Survival of silver maple within the replications was significantly higher than either cottonwood variety. The test between replications of survival within each cottonwood variety showed that the local cottonwood in replication 1, with no surviving trees, was significantly different than replications 2, 3, and 4, which were tested as not statistically different. For the Westvaco

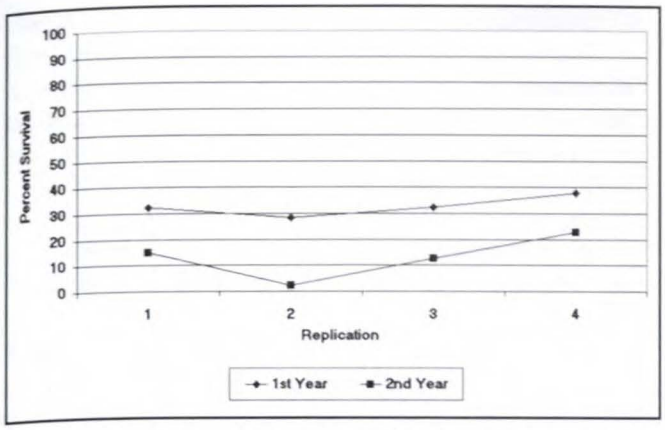

Figure 3.—*Percent survival of all species combined by replication.*

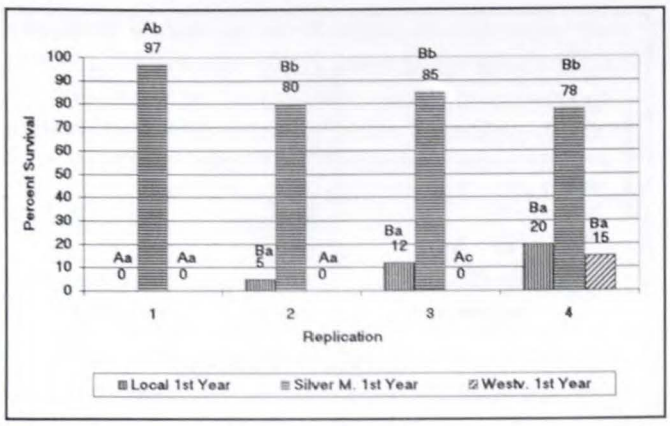

Figure 4.—*Percent survival for year 1 by replication and species. Different capital letters indicate significant differences (P=0.05) between percentage survival by species over replications; different lower case letters indicate significant differences between survival of species within replications.*

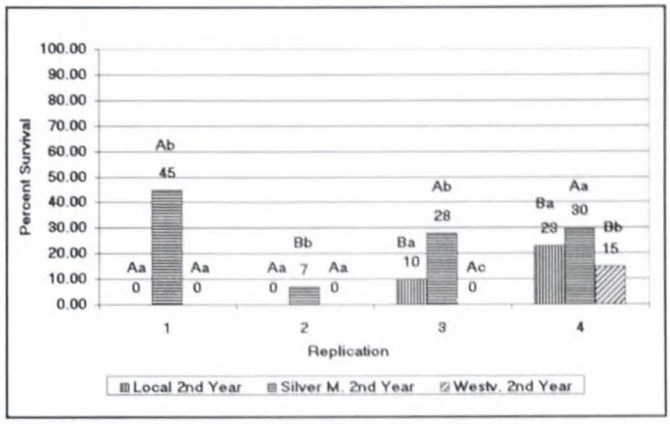

Figure 5.—Percent survival for year 2 by replication and species. Different capital letters indicate significant differences (P=0.05) between percentage survival by species over replications; different lower case letters indicate significant differences between survival of species within replications.

cottonwood that only had surviving trees in replication 4, there was a significant difference between the other replications. Yet, comparing the survival of both local and Westvaco cottonwood in replication 4, it was found that they were not statistically different.

By the end of year 2, survival comparisons by species across replication and species-to-species comparisons within replications had decreased (fig. 5). While silver maple still had higher survival in each replication, the number of surviving trees in replications 1, 3, and 4 were not significantly different. To summarize, in both years 1 and 2 silver maple had significantly better survival than either cottonwood selection, but the difference in percent survival had decreased by the end of year 2.

Species and Spacings Analysis

The hypothesis that survival did not change from year 1 to year 2 for each species was tested, and silver maple was the only species that had a statistically significant reduction in survival, from 85 percent in year 1 to 27.5 percent in year 2 (fig. 6). Though having low overall survival percentages, neither local nor Westvaco cottonwood varieties experienced a significant change in total survival between years 1 and 2.

First year survival comparisons between the 4 x 4 feet and 8 x 8 feet spacings showed a significant difference within both local and Westvaco cottonwood. Survival for local cottonwood showed a statistically significant increase from 5 percent on the 4 x 4 foot spacing to 13.3 percent on the 8 x 8 foot spacing (fig. 7). Although identified as significant, the trend was reversed for the Westvaco cottonwood that had 7.5 percent survival on the 4 x 4 foot spacing and 0 percent on the 8 x 8 foot spacing. Though not significant, the silver maple survival decreased from 85.8 percent (4 x 4 foot spacing) to 84.2 percent (8 x 8 foot spacing).

The influence of spacing treatment on species survival was significant. Survival of local cottonwood at the end of year 2 underwent a significant increase in survival, from 2.5 percent on the 4 x 4 foot spacing to 14.2 percent on the 8 x 8 foot spacing (fig. 8). The Westvaco cottonwood survival was significantly higher (7.5 percent) on the 4 x 4 foot spacing than on the 8 x 8 foot spacing (0 percent). By the end of year 2, silver maple survival had decreased to 22.5 percent on the 4 x 4 foot spacing and 32.5 percent on the 8 x 8 foot spacing, a difference that was not identified as significant.

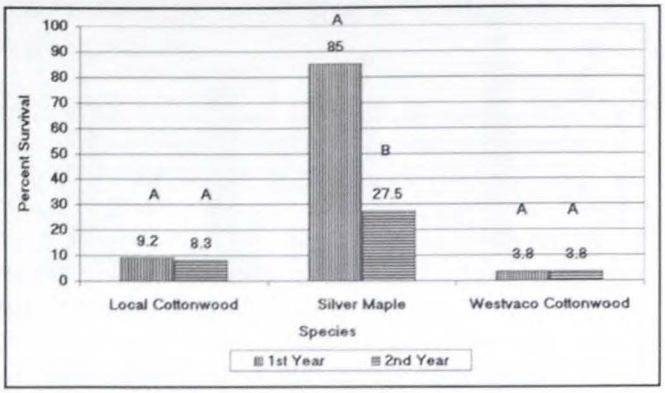

Figure 6.—*Percent survival of each species for years 1 and 2 Different capital letters indicate significant differences (P=0.05) between year 1 and year 2 means pooled over replications.*

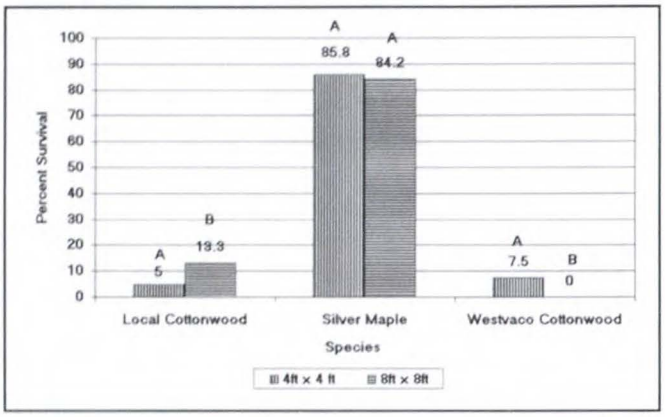

Figure 7.—*Percent live trees at the end of year 1 by species and spacing. Different capital letters indicate significant differences (P=0.05) between spacings for each species from means pooled over replications.*

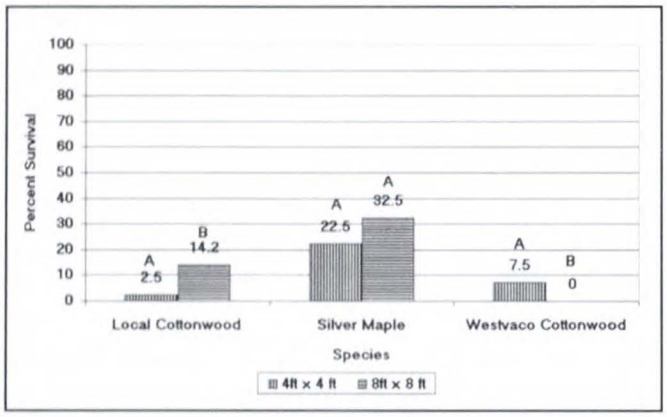

Figure 8.—Percent live trees by species and spacing at the end of year 2. Different capital letters indicate significant differences (P=0.05) between spacings for each species from means pooled over replications.

DISCUSSION

Survival

One of the reasons to use both a locally selected and superior cottonwood (Westvaco) was to provide a comparison based on the premise that the superior cottonwood clone would have greater growth than the local cottonwood. Silver maple was also used as a species that grows quickly and has the ability to coppice. Symonette (1988) and others have noted its potential for biomass utilization in products that use small wood particles. Both species should have been well matched to growing conditions in a floodplain environment.

By the end of the second year, the percentage survival by species and spacing was not high enough for an effective demonstration. Survival was at best poor. Silver maple had the best overall survival, yet total survival was only 27.5 percent (fig. 6). The poorest survival was Westvaco cottonwood at 3.75 percent and the local cottonwood was only slightly better at 8.33 percent survival.

In order to help understand the differences in total survival (all species combined) from year 1 to year 2 it is useful to look at the percentage survival by replication. The linear trend indicated that survival increased across the study site from replication 1 to replication 4 (fig. 9). It is likely that the survival trend lines in figure 9 are indicative of an increasing change in elevation from replication 1 to replication 4. Changes in elevation across all but the northwestern corner (fig. 2), were 1 percent or less. During the floods, measured depths of water across the replications indicated that the topography was hummocky due to uneven silt deposition. Though not consistent, these changes in elevation resulted in water depths as much as 2 feet deeper on replication 1 compared to replication 4.

The elevational change created a scenario whereby portions of replication 1 emerged from the floodwater as much as 2 weeks later than replication 4 (dependant in part on wind, solar radiation, and subsequent intervals between precipitation events). Therefore, it is likely that the trend in increased survival from replication 1 to replication 4 (fig. 9) reflects the difference in duration of submersion. It is also likely that submersion and the resultant decreased rates of photosynthesis, caused stunted growth in all tree species. Submersion of seedlings creates an oxygen-limiting situation that not only hinders their growth but may be fatal, also damage to

the leaves and root structures in response to flooding must affect tree growth (both height and diameter) which is dependent on photosynthesis and O_2 exchange (Bratkovich and others 1993, Colbert 1998, Kozlowski and Pallardy 1997, Peterson and Bazzaz 1984). The early emergence from floodwaters of replication 4, compared to replication 1, improved the survival of its seedlings.

Weed Control

In any agronomic system it is essential that competition for nutrients and water be eliminated. Similarly, in order to maximize the growth potential of a forest crop, competition must also be controlled. Given flooding dynamics of the area, the second year a cover crop was seeded in an effort to control weed competition. Neither accomplished the intended result. Herbicide was only partially effective and survival of the cover crop establishment was negatively affected by fall flooding.

In one respect the herbicide worked as intended. The silver maple had a good kill of the weeds within an effective zone of approximately 3-feet around each seedling. However, the weeds outside the spray zone grew so prolifically that they overtopped the growing space of each seedling. Also, by creating an open growing space around each seedling, there may have been a tendency to create a target for rodent damage. Many of the silver maple suffered from substantial rabbit damage, which resulted in a definite setback in growth.

The effectiveness of herbicide application was compromised by flooding which occurred four times during the establishment year (fig. 10). It is likely that these floods either washed out or diluted the preemergent herbicide from the soil. Flooding served to reestablish the previously eradicated weeds. Weed seed from areas adjacent to the site was visible, floating on top of waters over the research site. Flooding reduced our ability to control the weedy competition on the study site, which impacted the survival and ultimate biomass yield.

Flooding

Flooding was potentially the most influential factor affecting survival. Yet, the study design was not intended to test the effects of flooding. Though the influence of flooding was recognized at the onset of this project, the abnormal frequency and duration of the multiple floods could not have been foreseen as one of the most critical factors in survival.

During the establishment year the research site flooded on four separate occasions. Similarly, site flooding occurred in April, May, and the end of June during the 1999 growing season. This was documented using a river gauge located at Boonville, Missouri, which is the closest gauging station upriver from the research site. I observed and documented that at a Boonville river stage of 23 feet, water began to enter and cover the site (fig. 10). The flooding was a direct result of natural levee failures that occurred during the flood of 1993 and 1995. One of the main problems associated with this levee break was its proximity to the research site. The break is actually higher in elevation than the planting. Therefore, when water entered the site, it did not leave at the same rate. When water came onto the study site, it remained for approximately 30 days, dependent on subsequent rainfall, seasonal temperatures, and wind.

In retrospect then, one of the foremost considerations prior to planting in a floodplain

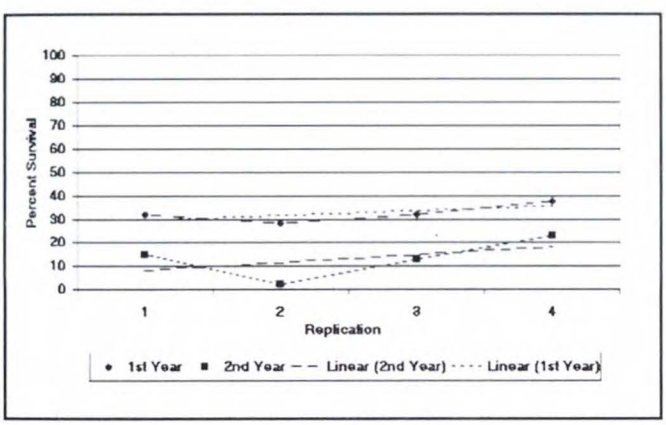

Figure 9.—*Percent survival for each replication by year with linear trend line plotted.*

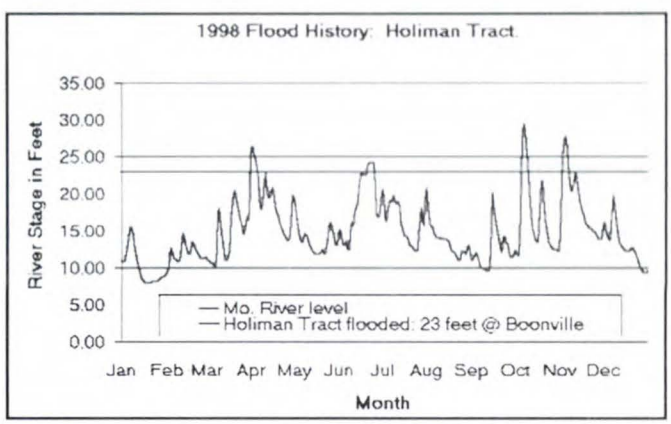

Figure 10.—*History of actual flood events on the Holiman tract for the calendar year 1998.*

should be to document its history of flooding. This should include the time of year and frequency of flooding events, as well as the duration flood waters remain on a specific site. Historic records were useful in determining the frequency of flooding over the past 10 years. Estimation of historic flooding was based on the observed flood level for the study site of 23 feet at Boonville and was used to create a history of flooding based on current natural levee and site conditions. Apart from the years 1993 and 1995, flooding most often occurred from February through July. Based on this data it might have been best to try fall planting on this site, though, for this site and the years of this study, that too would have failed. Raised-bed plantings with tall seedlings might also be a valid option to improve survival of trees established in floodplains.

CONCLUSION

The primary influence that detracted from this becoming an effective demonstration of the potential of SRWCs was the flooding and the associated mortality that took place. Yet, what we have learned from efforts to establish this agroforestry system in an unprotected floodplain will be useful in directing future attempts to utilize similar ground. One lesson we learned was that elevation plays an important role in survival during establishment. Floodplains by nature are relatively flat. But on large acreages of land a 1 percent change in topography can make a several foot difference in depth of floodwaters. Also, this most often equates to a shorter duration for submersion. Both depth and duration may mean the difference between the survival or death of a plantation.

Second, we learned it is also important to ascertain as much about the land and its flood history as possible. Past flood history may identify certain times of the year in which it is more likely that plantings will succeed. Finally, site preparation a year in advance may be necessary. The dynamics of these areas are so powerful that intensive work will be required far in advance of plantation establishment in order to successfully control competition.

ACKNOWLEDGMENT

The U.S. Environmental Protection Agency and the U.S.D.A. Agricultural Research Service, Dale Bumpers Small Farms Research Center, Booneville, AR funded this work. This research was also, in part, supported by the Missouri Agricultural Experiment Station project number PSSL 0112. The results presented are the sole responsibility of the Principal Investigators and may not represent the policies or positions of the funding agencies.

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COMPARISON OF SITE PREPARATION METHODS AND STOCK TYPES FOR ARTIFICIAL REGENERATION OF OAKS IN BOTTOMLANDS

Gordon W. Shaw, Daniel C. Dey, John Kabrick, Jennifer Grabner, and Rose-Marie Muzika[1]

ABSTRACT.—Regenerating oak in floodplains is problematic and current silvicultural methods are not always reliable. We are evaluating the field performance of a new nursery product, the RPM™ seedling, and the benefit of soil mounding and a cover crop of redtop grass to the survival and growth of pin oak and swamp white oak regeneration on former bottomland cropfields. First-year survival was high for both RPM™ and bareroot seedlings. Basal diameter increment was significantly greater for RPM™ seedlings than bareroot stock. Although height increment was greater for RPM™ than bareroot, differences were insignificant. Neither soil mounding nor cover crop treatments significantly affected oak performance in the first year.

In Midwestern floodplains, oaks (*Quercus* sp.) and other nut (hard mast) species such as hickory and pecan (*Carya* sp.) and black walnut (*Juglans nigra* L.) were members of diverse bottomland forests containing cottonwood (*Populus deltoides* Marsh.), silver maple (*Acer saccharinum* L.), hackberry (*Celtis occidentalis* L.), sycamore (*Platanus occidentalis* L.), and black willow (*Salix nigra* Marsh.). The oaks and other nut species occurred on higher elevations that were flooded less frequently and had better-drained soils.

Oak species, including swamp white oak (*Quercus bicolor* Willd.), bur oak (*Q. macrocarpa* Michx.), and pin oak (*Q. palustris* Muenchh.), were present in the Lower Missouri and Upper Mississippi River floodplains based on the known geographic distribution and silvics of native tree species and on analyses of General Land Office surveys conducted in the early 1800s (Bragg and Tatschl 1977, Burns and Honkala 1990, Nelson 1997, Yin and others 1997, Nelson and others 1998). In Missouri, oaks were common enough to be recorded on about one-third of the survey transects of the Missouri River floodplain in the early 1800s (Bragg and Tatschl 1977). Elsewhere, oak-hickory was a recognized floodplain forest type in the Upper Mississippi River basin before European settlement. Oaks were also associates in other floodplain forest and savanna types.

Public land managers and private landowners are interested in reforesting former agricultural bottomlands with a component of native oaks and other hard mast producing tree species for:

1) improving wildlife habitat,
2) promoting native biodiversity,
3) providing recreation,
4) enhancing waterfowl hunting,
5) producing forest products, and
6) restoring ecosystem process and function.

Managers are using both natural regeneration methods and artificial regeneration techniques (tree planting and direct seeding) to meet reforestation objectives in a timely manner.

Natural regeneration is the preferred method for restoring bottomland forests because it is the least costly regeneration method. Natural regeneration of pioneer species such as cottonwood, sycamore, and willow is common following abandonment of bottomland crop fields because

1) their seed sources are widely distributed throughout the floodplain,
2) they frequently produce good seed crops,

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3) their light seeds are readily dispersed by wind and water,
4) they germinate well on mineral soil seedbeds, and
5) they have rapid initial shoot growth.

In contrast, the natural development of later successional bottomland species such as swamp white oak, bur oak, pin oak, black walnut, pecan, and shellbark hickory (*Carya laciniosa* (Michx. F.) Loud.) is more problematic due to a lack of seed source and their low regeneration potential on productive sites that are often dominated by lush herbaceous growth, woody vines, and pioneer tree species that exhibit inherently greater shoot growth. Therefore, artificial regeneration and vegetation management is typically required to reestablish species such as the oaks in floodplain environments.

However, attempts to establish oak and other hard mast producing tree species in bottomland fields using artificial regeneration methods such as direct seedling and planting of bareroot seedlings have often failed. For example, in a survey of 4-year-old Wetland Reserve Program plantings in the Mississippi River floodplain, Schweitzer and Stanturf (1997) found that only 9 percent of the total reforested land in 13 Mississippi counties met the Natural Resources Conservation Service requirement for at least 125 hard mast stems per acre in 3-year-old stands.

High rates of tree mortality and slow growth can result from
1) the use of poor quality seed or seedlings;
2) improperly planted seedlings, or sown seed;
3) damage caused by deer, rabbits, and mice;
4) drought;
5) severe competition; and
6) flooding (Stanturf and others 1998).

Loss to flooding is higher when species are planted off-site in the floodplain, i.e., their flood tolerance is not matched with the hydrologic regime of the site. Slow growing and damaged seedlings are easily overtopped by competing vegetation on these productive sites. Even when planting has been done properly, regenerating oaks in productive floodplains has proven difficult because of their slow juvenile growth habit and their adaptations to specific soil and microtopographic environments within floodplains. Further complications in regenerating oaks arise because many natural fluvial processes have been altered, changing soil, microtopographic, and hydrologic patterns.

There are a number of stock type choices and cultural methods that may improve the survival and growth of planted seedlings in bottomlands. Seedlings with larger initial diameters and more fully developed root systems have greater survival and growth rates (Kormanik and others 1995, 1998) and therefore may be more competitive on productive bottomlands. Cover crops can be planted to suppress weedy competition as well as improve seedling survival and growth (Van Sambeek and others 1986, Alley and others 1999). Soil mounding improves seedling survival and growth by raising seedlings above flood waters or high water tables, removing excess surface water, decreasing bulk density, increasing aeration, and concentrating organic matter in the root zone (Fisher and Binkley 2000).

The purpose of this study is to learn how to regenerate pin oak and swamp white oak on former agricultural bottomlands along the Lower Missouri River. We are evaluating how cultural methods such as planting trees on a soil mound and controlling competition with a cover crop can be used with different nursery seedling types to more successfully regenerate oak by improving its competitiveness in early successional bottomland communities. We are testing the field performance of a new nursery product, the RPM™ seedling whose performance will be compared with that of the more traditional 1-0 bareroot seedlings. Oak regeneration success will be related to vegetation competition, composition, and structure under the different silvicultural prescriptions. Our research focuses on solving the common problem of regenerating oaks in bottomlands and hence will contribute to our ability to reforest bottomlands with native oaks. We recognize that reestablishing oaks and other individual tree species is just one small component of restoring native bottomland forests.

METHODS

Study Area

The study is located on two conservation areas managed by the Missouri Department of Conservation: Smoky Waters (Sec. 5, T 44 N, R 9 W and Sec. 1, T 44 N, R 10 W; Cole County) and Plowboy Bend (Secs. 24 and 25, T 47 N, R 14 W; Moniteau County). Soils at the Plowboy Bend site were mapped as Sarpy Fine Sand (mixed, mesic, Typic Udipsamments). Soils at the Smoky Waters site were mapped as Haynie Silt Loam (coarse-silty, mixed, superacitve, calcarious, mesic Mollic Udifluvents) and Leta Silty

Clay (clayey over loamy, smectitic, mesic, Fluvaquentic Hapludolls). Haynie and Sarpy soils are classified as hydric, meaning that they are periodically saturated, ponded, or flooded with water during the growing season.

Plowboy Bend is protected from the Missouri River by a levee; Smoky Waters is not. The sites have not experienced flooding since the beginning of the study, although both sites were inundated during the Great Flood of 1993. Crops were previously grown in these fields until the fall of 1998. By the late summer of 1999, all study fields had a cover of herbaceous weeds and native plants.

Design

We used a split plot design for the study. We established six 40-acre fields equally divided between Smoky Waters and Plowboy Bend Conservation Areas. Each field is laid out as a square with a side being 1,320 feet in length. One of the 40-acre fields (fig. 1) at each conservation area has received no treatment after the initial site preparation, which is described below. This field represents the control and will be monitored to document natural succession that follows abandonment of bottomland cropfields. The remaining two fields at each site were designated as "management" fields and were each randomly assigned a cover crop treatment: redtop grass (*Agrostis gigantea* Roth) cover crop or no cover crop. The no cover crop field allows competing vegetation to develop naturally with the planted oak seedlings.

Within each of the two management fields, soil-mounding treatments (mounded or not) were randomly assigned across each field in groups of five rows each (fig. 2). Rows were spaced 30 feet apart and oriented parallel to the Missouri River to ensure that soil mounds would not impound surface waters. Within each group of five rows, we designated planting units to be five rows wide and 150 feet long and randomly assigned one combination of species and stock type to each of them. Each planting unit contained 30 planting sites.

Stock types included 1-0 bareroot, and two classes of RPM™ seedlings. RPM™ is a trademark for the Root Production Method, an air root pruning process developed by Forrest Keeling Nursery in Elsberry, MO (Lovelace 1998). This nursery culture technique produces a large container-grown seedling that has a dense, fibrous root system. Trees are grown in 3- or 5-gallon containers and attain heights ≥ 5 feet tall in 1 to 2 years in the nursery. In our study, RPM™ seedlings were either 1.5-year-old trees in 3-gallon pots (RPM3), or 2-year-old trees in 5-gallon pots (RPM5). Species being tested are pin oak and swamp white oak.

Seedlings were planted on a spacing of 30 feet within each row (30 x 30 foot spacing = 48 trees/ac). Within a cover crop field, mounding, stock type, and species treatments were replicated four times (fig. 3). Additional trees were planted at field edges to reforest the entire 40 acres.

Figure 1.—*Design of study treatment fields at Plowboy Bend Conservation Area. Each field is 40 acres in area and contains 44 rows on 30-foot centers. Rows are either mounded or not. The no management field is the control where no efforts were made to direct succession.*

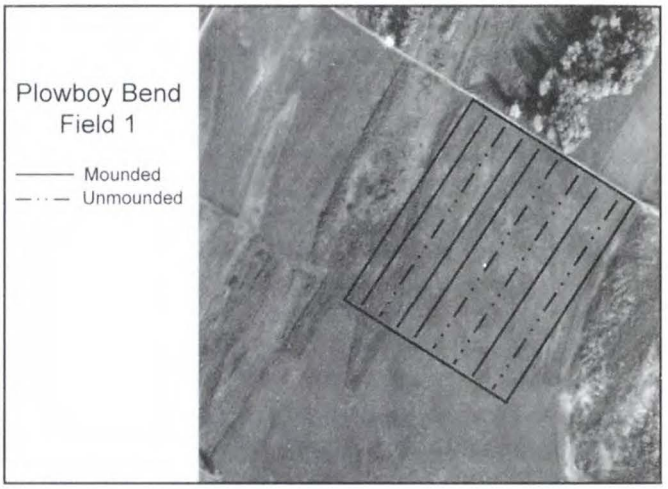

Figure 2.—*Soil mounding treatments were randomly assigned across each 40-acre management field. Each line represents 5 rows on 30-foot centers.*

Average root volumes for pin oak and swamp white oak bareroot seedlings were 1.59 in^3 and 2.03 in^3, respectively (table 1). Average root volumes for RPM3 and RPM5 pin oak seedlings were 14.44 in^3 and 13.58 in^3, respectively while those for swamp white oak RPM3 and RPM5 seedlings were 8.62 in^3 and 15.39 in^3, respectively. Root dry weight, shoot length, and shoot dry weight were also greater for RPM trees than for the bareroot stock.

Before establishing the experiment, we sampled soils to identify patterns in soil texture, hydrology, and other physical properties that may influence plot layout. Soil texture and drainage regime, as evidenced by soil morphology, were determined to a depth of 4 feet using an auger at both Plowboy Bend and Smoky Waters. We sampled soils systematically across all fields, but also sampled microtopographic features such as ridges, old side channels, and other shallow depressions. Microtopography, soil texture, and drainage did not vary substantially over the study areas and we concluded it would not affect the location or design of treatment plots.

Site Preparation and Establishment

The Plowboy Bend study site was mowed with a tractor and brush-hog in July of 1999, and gone over twice in perpendicular directions with an offset disk in August to prepare it for tree planting and sowing of the redtop grass. Site preparation at the Smoky Waters study site differed somewhat from that at Plowboy Bend because plant biomass was considerably greater. At Smoky Waters, herbicides (2,4-D and Roundup®) were applied with an agricultural

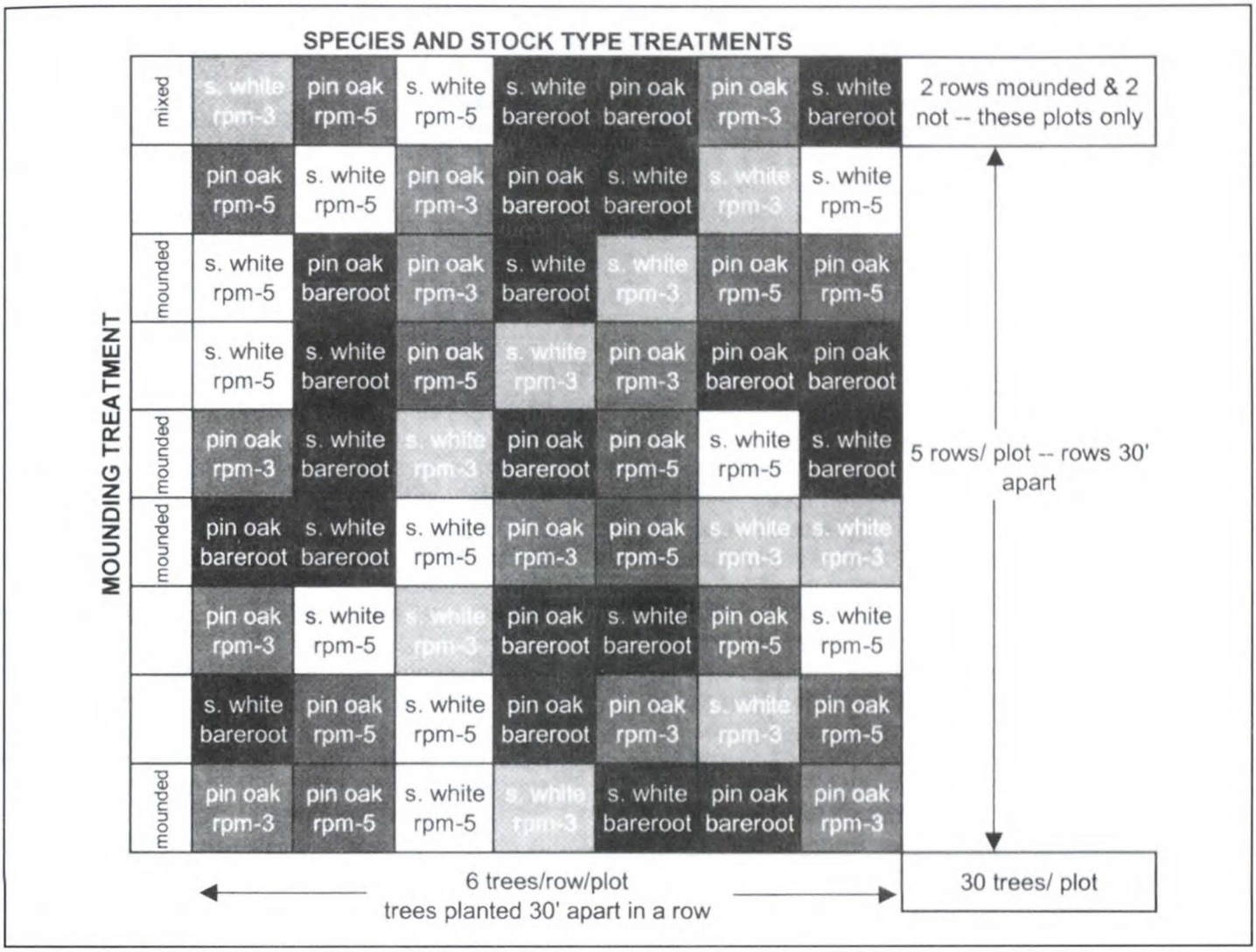

Figure 3.—*Layout of Smoky Waters no-redtop treatment field. The various combinations of species and stock types were randomly located along the soil mounding treatment rows and were replicated 4 times within each 40-acre management field. Additional plantings were made to fill out the 40 acres. rpm-3 = RPM™ seedling in a 3-gallon container. rpm-5 = RPM™ seedling in a 5-gallon container.*

Table 1.—*Measurements of randomly selected seedlings by species and stock type at time of planting (BR = bare-root seedling; RPM3 = RPM™ seedling in a 3-gallon container; RPM5 = RPM™ seedling in a 5-gallon container). Basal diameter was measured 1 inch above the root collar (mean ± standard deviation).*

Oak species	Stock type		Basal diameter	Root volume	Root dry weight	Shoot length	Shoot dry weight
		(#)	(inches)	(in.^{3} H_2O displaced)	(grams)	(inches)	(grams)
Pin	BR	30	0.30 ± 0.05	1.59 ± 0.49	16.7 ± 5.9	27.5 ± 3.6	9.85 ± 3.3
Pin	RPM3	27	0.83 ± 0.14	14.44 ± 5.88	117.2 ± 47.8	92.5 ± 15.8	215.8 ± 83.4
Pin	RPM5	34	0.78 ± 0.15	13.58 ± 5.69	118.2 ± 42.9	91.4 ± 10.2	187.3 ± 65.2
Swamp white	BR	30	0.30 ± 0.05	2.03 ± 0.67	20.5 ± 5.98	17.9 ± 2.61	7.70 ± 2.98
Swamp white	RPM3	31	0.58 ± 0.08	8.62 ± 3.59	63.7 ± 23.5	58.2 ± 9.80	80.0 ± 31.7
Swamp white	RPM5	29	0.76 ± 0.12	15.39 ± 5.42	137.7 ± 48.5	82.8 ± 11.2	195.7 ± 69.4

Table 2.—*Initial seedling size measured by species and stock type (BR = bare-root seedling; RPM3 = RPM™ seedling in a 3-gallon container; RPM5 = RPM™ seedling in a 5-gallon container). Basal diameter was measured 1 inch above the ground surface (mean ± standard deviation).*

Oak species	Stock type	Sample size	Basal diameter	Height
		(# trees)	(inches)	(feet)
Pin	BR	1,175	0.23 ± 0.05	1.1 ± 0.2
Pin	RPM3	1,204	0.75 ± 0.13	7.2 ± 1.2
Pin	RPM5	1,269	0.79 ± 0.14	7.6 ± 1.2
Swamp white	BR	1,192	0.25 ± 0.06	1.1 ± 0.2
Swamp white	RPM3	1,274	0.66 ± 0.13	5.8 ± 1.1
Swamp white	RPM5	1,248	0.66 ± 0.13	6.1 ± 1.1

boom-type sprayer in early August. The herbicide was applied at a rate of 2.0 qt/ac of Roundup, 1.0 qt/ac of 2,4-D and 2 lbs/ac of water-soluble ammonium sulfate. Within a month, the study site at Smoky Waters was disked the same way as was done at Plowboy Bend.

In mid-September of 1999, the redtop fields were worked with an offset disk, and soil mounds were constructed with a levee plow (AMCO LF6-824) on all but the control fields. Soil mounds were about 2 feet in width at the top of the berm and 7 feet at the base. After the soil settled, mounds were about 12 inches to 15 inches in height above the natural ground elevation. In September, immediately following disking and soil mounding, redtop grass was sown at 6 lbs/ac with a two-gang roller (Brillion) seeder.

For each oak species, approximately 1,200 seedlings of each stock type were planted in randomly located 30-tree plots (fig. 3), for a total of 7,362 trees over the entire study area. RPM™ trees were planted in November of 1999 and the 1-0 bareroot seedlings were planted in the spring of 2000. Heights and basal diameters were similar between the two RPM™ stock types for each oak species. Height averaged 7.2 feet and 7.6 feet for pin oak RPM3 and RPM5 trees, respectively (table 2). Regardless of RPM™ stock type, pin oak seedlings averaged 0.8 inches in basal diameter. Swamp white oak RPM™ seedlings were slightly smaller in height and basal diameter than the pin oaks. In contrast, bareroot seedlings averaged about 1.1 feet in height and 0.2 inches in basal diameter, regardless of species.

At the time of planting, a slow release fertilizer (33-3-6 NPK) was applied to the ground surface around each seedling at an approximate rate of 2 oz. per tree. A 4 x 4 foot woven plastic weed mat was placed around each seedling in the spring of 2000.

Measurements and Analyses

Soil temperature and water were monitored at 4-inch and 12-inch depths in both mounded and non-mounded planting rows to determine if and how soil mounding affects the planted seedlings. Measurements were made in adjacent mounded and non-mounded rows at random locations within each 40-acre field. Data included in this paper are from single, randomly located installation pairs in each planted 40-acre field. Soil temperature was recorded every 3 hours with HOBO® H8 Outdoor/Industrial data loggers and permanently installed TMCx-HA temperature sensors (both from Onset Computer Corporation, Pocasset, MA).

Soil water was measured weekly with permanently installed Watermark Soil Moisture Sensors (Irrometer Company, Inc., Riverside, CA). Soil temperature data were used to adjust the Watermark Sensor meter prior to each measurement. To convert Watermark Sensor meter readings to gravimetric water content, we conducted a laboratory calibration with soil samples collected at each sensor installation and developed conversion factors for each sampling location.

To further characterize the soils at each study site, soil pits were excavated near each temperature and water sensor installation in both mounded and non-mounded locations for soil morphology description and laboratory analysis. Excavations were at least 60 inches deep. Soil horizons were identified and described and their thicknesses were recorded. Horizon description included soil color, structure and consistence, root and pore sizes and abundances, and the presence and color of redox features.

Soil samples from each soil horizon were collected for analysis at the University of Missouri Soil Characterization Laboratory including particle size distribution, extractable cations, cation exchange capacity, base saturation, organic carbon content, and pH. All soil samples were air dried and passed through a 0.08 in. sieve to obtain the fine earth fraction. Particle size distribution was determined by pipette (Gee and Bauder 1986). Cations were extracted with 1M-ammonium acetate at pH 7.0; K was determined by flame emission, and Ca and Mg by atomic absorption. Total carbon was determined by dry combustion with a Leco CR 12 Carbon Analyzer (Leco Corp., St. Joseph, MI) and pH was measured in a 1:1 solution suspension. Data from selected soil pits from each site are shown in table 3.

Initial total height and basal stem diameter 1 inch above the ground were measured on all seedlings after planting before the growing season and again at the end of the first growing season from September 2000 through April 2001.

From June through August of 2000, ground layer vegetation was monitored using randomly located paired 11-ft^2 quadrats and 33 ft x 66 ft macroplots, which were permanently marked to facilitate remeasurement. Percent ground cover by species was determined on the quadrats and macroplots. Paired quadrats, one between seedlings within a planted row and one located perpendicular to it between the rows, were used to quantify composition and abundance of competing vegetation within treatment plots. Within each 40-acre field, three replications of mounding, stock type, and species treatment combinations were sampled with paired quadrats. A total of seventy-two 11-ft^2 quadrats were installed in each field.

Three macroplots per 40-acre field were established to more thoroughly document total plant species diversity and successional patterns over time. Vertical structure of the ground flora was quantified at each quadrat location between the rows using a density board to estimate percent cover of grasses, woody vegetation, forbs, and total vegetation by 0.80-foot height increments up to a height of 6.6 feet (Hays and others 1981).

By summing the number of living stems and dividing by the total number originally planted we determined first-year survival. First-year height and diameter increments were calculated by subtracting initial measurements from the measurements made after the first growing season. We used analysis of variance (ANOVA) to determine significant ($P<0.05$) growth rate differences among species and stock type combinations between fields with and without redtop grass, and between mounded and non-mounded planting units. In this ANOVA, we used site as a random effect and tested cover crop treatment, mounding treatment, and species and stock type combinations as fixed effects with their respective site interactions as error terms. Because we found significant differences in basal diameter growth by species and stock type by site, we conducted a Duncan multiple range comparison test.

Table 3.—*Soil properties from excavations at each research site*

Location Horizon	Depth	Organic carbon	Sand	Silt	Clay	Texture class	Ca	Mg	K	CEC	pH
	in.	%	%	%	%		cmol(+)kg^{-1}				
PLOWBOY BEND											
Ap1	0-6	0.4	84	13	3	Loamy sand	9.5	1.6	0.3	5.1	7.7
Ap2	6-10	0.4	75	20	5	Loamy sand	7.2	2.1	0.3	6.3	7.2
Cg1	10-17	0.3	51	43	6	Sandy loam	14.4	2.4	0.3	7.8	7.2
Cg2	17-3	0.4	28	63	9	Silt loam	18.4	2.0	0.3	10.3	7.5
Cg3	23-29	0.3	38	53	9	Silt loam	19.1	2.0	0.3	9.6	7.5
C1	29-43	0.1	90	8	2	Sand	11.4	1.2	0.1	3.2	7.6
C2	43-60+	0.1	94	5	1	Sand	7.6	0.8	0.1	2.6	7.7
SMOKY WATERS											
Ap1	0-5	1.3	13	56	31	Silty clay loam	28.7	4.8	1.0	24.8	8.0
Ap2	5-8	0.7	39	45	16	Loam	16.5	2.9	0.4	13.6	8.0
AB	8-13	1.1	2	67	31	Silty clay loam	19.6	4.4	0.5	25.4	8.0
Bg1	13-18	1.1	2	60	38	Silty clay loam	22.4	5.2	0.6	29.7	7.8
Bg2	18-24	1.0	2	58	40	Silty clay	21.7	6.0	0.8	29.6	7.7
Bg3	24-28	0.7	3	58	39	Silty clay loam	24.3	6.1	0.7	29.1	7.8
Cg1	28-34	0.4	1	78	21	Silt loam	22.2	4.4	0.4	18.0	7.9
Cg2	34-42	0.3	3	83	14	Silt loam	21.2	2.8	0.3	13.2	8.0
Cg3	42-49	0.3	8	79	13	Silt loam	19.8	2.4	0.3	12.0	7.9
Cg4	49-60+	0.4	28	63	9	Silt loam	20.0	2.0	0.3	11.1	8.0

RESULTS

Soil Water

Gravimetric soil water content at the 4-inch depth was greater at Smoky Waters (averaging 27 percent) than at Plowboy Bend (averaging 13 percent). At each site, soil water content was lower in the redtop field than in the no-redtop field and average differences ranged from 3 to 5 percent at Smoky Waters and 7 to 10 percent at Plowboy Bend. At the 4-inch depth, soil mounding decreased soil water content by an average of 3 percent at both Smoky Waters and Plowboy Bend (table 4). However, the effect of soil mounding on soil water content at the 12-inch depth differed by site. At Smoky Waters, soil mounding decreased soil water content by an average of 5 percent compared to 2 percent at Plowboy Bend.

Although soil water content differences between mounded and non-mounded soils were small when averaged for a growing season or an entire year, we observed that mounded soils were 10 to 25 percent drier than non-mounded soils shortly after rainfall (fig. 4). Greater soil water differences were observed at the 12-inch depth than at the 4-inch depth.

Vegetation Development

During the summer of 2000, we identified 101 plant species across all study fields at Smoky Waters and Plowboy Bend Conservation Areas. Most plants were annual and biennial forbs and grasses, of which slightly more than half were exotic species. Common native species included purslane speedwell (*Veronica peregrina* L), cutleaf primrose (*Oenothera laciniata* Hill), and horseweed (*Conyza canadensis* L.). Many species (N=81) were not widely distributed, occurring on less than 10 percent of all the 11-ft^2 quadrats. Woody species inventoried included boxelder (*Acer negundo* L.), eastern cottonwood, hackberry, and trumpet creeper (*Campsis radicans* L.). Mean percent ground cover, in general, was greater at Smoky Waters than that at Plowboy Bend (fig. 5).

Sowing redtop grass as a cover crop increased the total percent ground cover at all sites and reduced the percent cover of other herbaceous species (fig. 5). At Smoky Waters, mean percent ground cover of other plants was reduced from about 52 percent on the no-redtop field to about 15 percent on the redtop field. Similar reductions were observed at Plowboy Bend. Species richness (mean number of species per 11 ft^2 plot) was slightly higher at Plowboy Bend than Smoky Waters.

Table 4.—*Seasonal average gravimetric water content for the two research sites at two different soil depths and by cover crop and mounding treatments*

Location and season	Soil depth	NO REDTOP		REDTOP	
		Mounded	Non-mounded	Mounded	Non-mounded
	in	% (wt/wt)	% (wt/wt)	% (wt/wt)	% (wt/wt)
PLOWBOY BEND					
Winter 2000	4	17	21	8	8
	12	18	20	11	11
Spring 2000	4	12	14	4	6
	12	10	17	3	8
Summer 2000	4	17	24	11	11
	12	22	28	31	12
Fall 2000	4	18	23	6	8
	12	16	20	7	9
Winter 2001	4	3	10	2	3
	12	11	13	4	5
Spring 2001	4	33	35	6	11
	12	9	23	2	8
Summer 2001	4	16	24	9	18
	12	10	18	2	8
Overall average	4	16	21	7	9
	12	15	20	11	9
SMOKY WATERS					
Winter 2000	4	31	29	24	30
	12	34	35	32	39
Spring 2000	4	24	27	16	21
	12	23	29	16	22
Summer 2000	4	25	27	24	28
	12	25	30	16	18
Fall 2000	4	34	35	26	30
	12	29	38	26	36
Winter 2001	4	12	31	9	11
	12	31	37	22	29
Spring 2001	4	Missing data	32	20	24
	12	26	34	17	26
Summer 2001	4	Missing data	20	32	34
	12	21	23	27	19
Overall average	4	27	28	22	27
	12	27	32	22	27

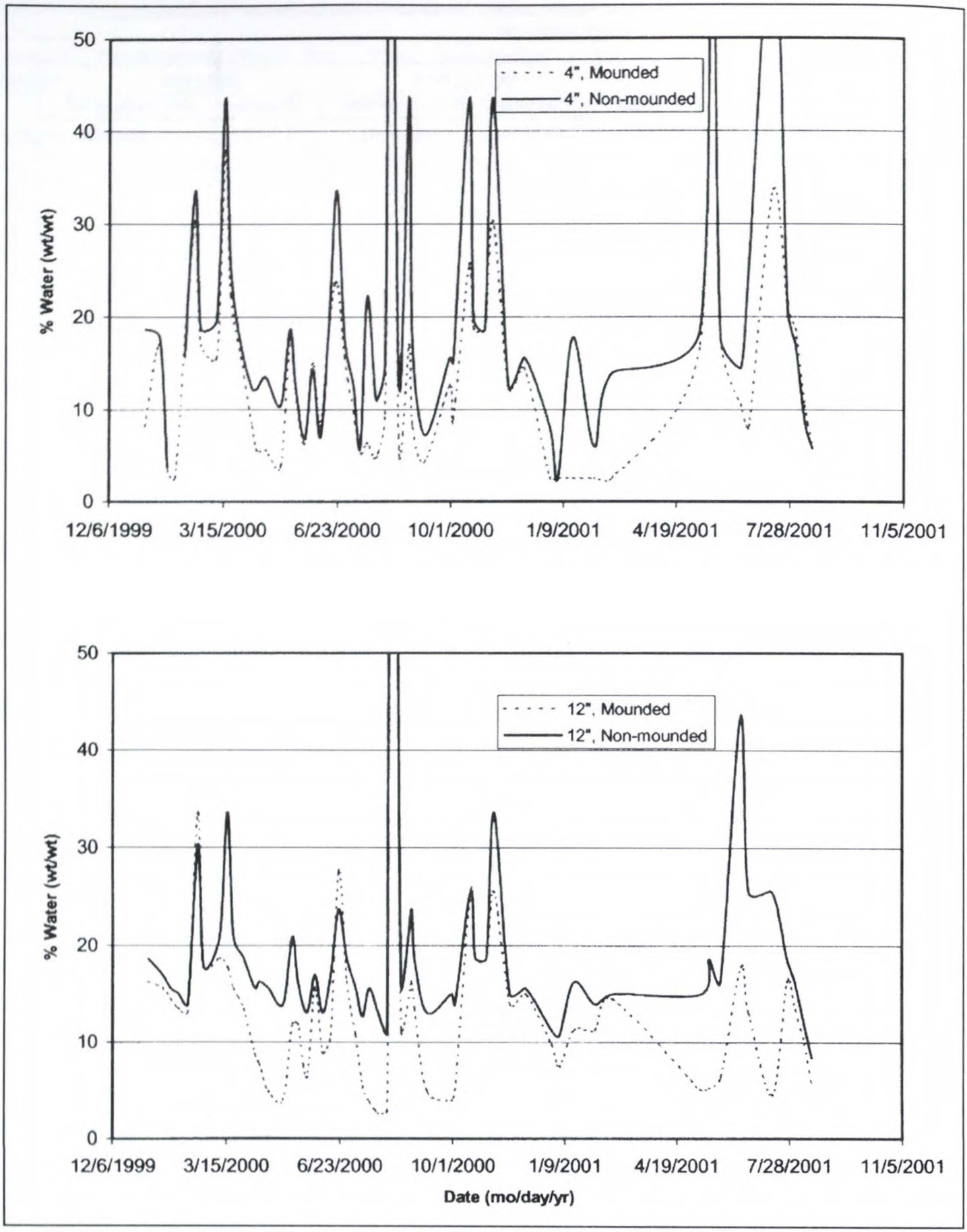

Figure 4.—*Percent soil water in mounded and non-mounded soils at 4-inch and 12-inch depth through time at Plowboy Bend Conservation Area.*

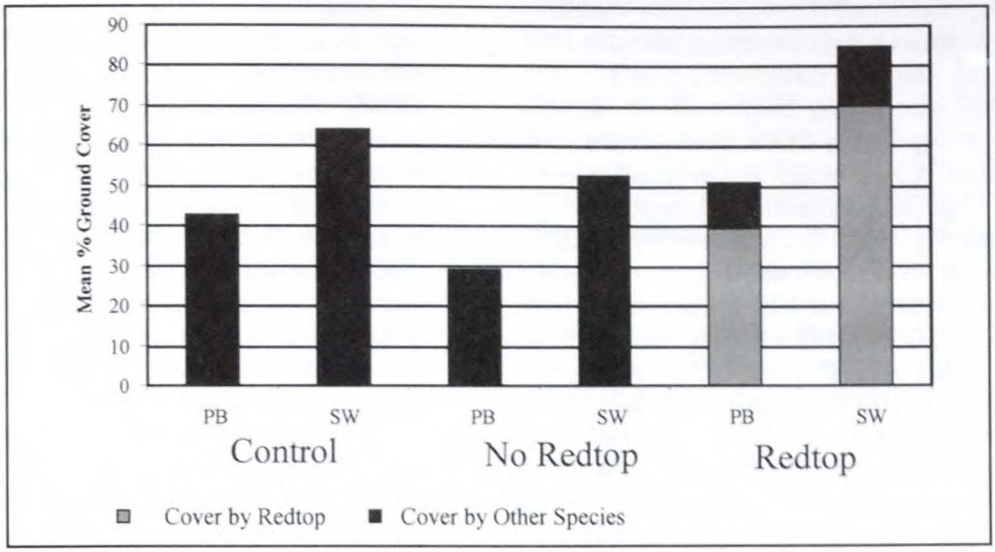

Figure 5.—*Mean percent ground cover (≤ 1.0 m above ground) by cover crop treatment at Plowboy Bend (PB) and Smoky Waters (SW) Conservation Areas.*

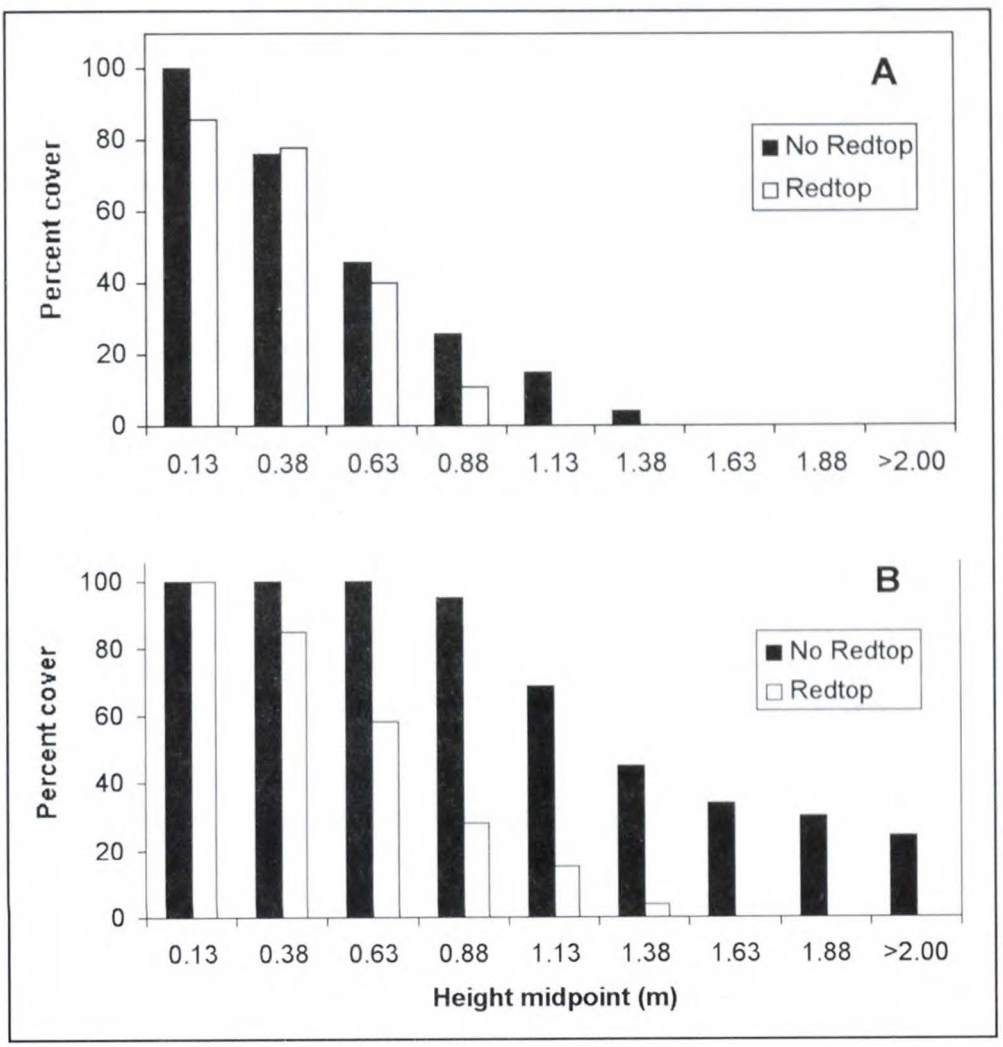

Figure 6.—*Vertical structure of ground flora at Plowboy Bend (A) and Smoky Waters (B) by cover crop treatment.*

Height and composition of the vegetation varied by cover crop treatment and between the two study sites (fig. 6). Percent cover decreased more rapidly with increasing height above the ground on the redtop grass fields than on the no-redtop fields at both Smoky Waters and Plowboy Bend. Vegetation reached greater heights at Smoky Waters than at Plowboy Bend regardless of cover crop treatment. At Plowboy Bend, vegetation grew to 4.9 feet on the no-redtop field and 3.3 feet on the redtop field. At Smoky Waters, vegetation grew to heights exceeding 6.6 feet on the no-redtop field but little vegetation was taller than 3.3 feet on the redtop field. On the no-redtop fields at Smoky Waters, total percent cover was substantial above 3.3 feet, and it averaged 24 percent above 6.6 feet. Grasses dominated (50 to 65 percent of the total percent cover) the vegetation up to 2.5 feet on the redtop fields, but they made up little of the vegetation on the no-redtop fields at both study sites.

Oak Seedling Survival and Growth

Survival was high for all species and stock types (table 5). Both RPM3 and RPM5 seedling survival was approximately 99 percent. Bareroot pin oak survival was 81 percent and bareroot swamp white oak survival was 93 percent. There was no statistical difference in seedling height growth. Bareroot seedlings showed negligible or negative height increment due to shoot dieback and resprouting, a common problem that is observed during the establishment period for hardwood bareroot seedlings (e.g., Johnson 1984). Average first-year height increment ranged from 0.3 feet for RPM3 pin oaks to 0.5 feet for RPM5 swamp white oak and pin oak seedlings.

The RPM™ seedling basal diameter growth was significantly greater ($P<0.001$) than for bareroot seedlings (table 5). Basal diameter growth of RPM™ seedlings was 0.09 to 0.16 inches, whereas bareroot growth was < 0.03 inches regardless of species (table 5). We found no significant differences in survival and growth of seedlings on mounded and non-mounded planting sites. We also found no significant differences in seedling survival and growth among fields with and without the redtop cover crop, but we recognize that this comparison lacked statistical power because of the small sample size.

DISCUSSION

Greater soil water content at Smoky Waters was caused by its finer-textured soil, which has a greater water-holding capacity than does the sandier soil at Plowboy Bend. This greater soil water content at Smoky Waters is one reason why there was greater vegetation ground cover and height than at Plowboy Bend. However, we are not certain why the redtop fields at both sites each had lower soil water contents than no-redtop fields. Although we initially suspected that the redtop grass and other vegetation in the redtop fields transpired more water than the vegetation in the no-redtop fields, we could not be certain of this because soil water content differences between redtop and no-redtop fields also occurred during the winter when vegetation was dormant. We checked our soil data to see if soil water differences could be attributed to differences in soil texture but found that soil textures at the water sensor installations did not appreciably differ from each field. However, there may be other site-specific factors causing these soil water content differences between the redtop and no-redtop fields.

Table 5.—*Summary statistics of first-year survival and growth of oaks planted in former agricultural crop fields in the Missouri River floodplain. (BR = bareroot seedling; RPM3 = RPM™ seedling in a 3-gallon container; RPM5 = RPM™ seedling in a 5-gallon container). Numbers within a column having the same superscript letter are not statistically different.*

Oak species	Stock type	Percent survival	Mean height increment (ft)	Mean basal diameter increment (in.)
Pin	BR	81	-0.1[a]	0.01[a]
Pin	RPM3	99	0.3[a]	0.12[b]
Pin	RPM5	100	0.5[a]	0.16[c]
Swamp white	BR	93	0.0[a]	0.03[d]
Swamp white	RPM3	99	0.4[a]	0.09[e]
Swamp white	RPM5	99	0.5[a]	0.11[b]

Although the soil mounding has not enhanced seedling survival or growth to date, it does appear to be decreasing soil water content. Soil water content differences between mounded and non-mounded soils were small when averaged by season or by year. However, mounded soils drained more quickly than non-mounded soils after rainfall. The full benefits of soil mounding may not become apparent until the sites experience flooding.

After 1 year, annual and biennial forbs and grasses dominated the plant community on these former bottomland crop fields. Exotic species comprised more than half of the species present during the first summer. The establishment of a cover crop of redtop grass greatly modified the development of vegetation on former bottomland crop fields after 1 year. Sowing redtop grass increased the total percent ground cover, effectively reduced the presence and vertical canopy coverage of other competing vegetation, decreased the height of competing vegetation to less than 3.3 feet, and increased the importance of grasses in the plant community. Grasses had an insignificant presence on the no-redtop fields, where competing vegetation grew to heights of 6.6 feet and higher. The vertical structure of the ground flora was more complex and well developed on the fields without redtop grass.

Total percent cover was greater with increasing heights without redtop grass, and maximum development occurred at Smoky Waters where there was still more than 20 percent total canopy cover in the 6.6 feet and above height category. Although there were substantial differences in plant community composition and structure between the cover crop treatments and between the two study sites, it is too early to conclude that the redtop grass competes less with planted oaks than does the mixture of herbaceous vegetation that develops in the absence of redtop grass. However, in the floodplains, where water is usually plentiful, light may be the limiting factor. Planting a cover of redtop grass may be a way of reducing seedling competition for sunlight. Redtop grass may also be valuable as a control of seedling damage by rodents because it reduces the escape cover available to rodents near the seedlings. This could reduce rodent populations and rodent browsing of seedlings.

The two RPM™ stock types were similar in their field performance regardless of species. Soil mounding and the use of a cover crop did not increase seedling survival or growth during the first year after planting. Survival was very high, even for the bareroot seedlings. Basal diameter growth of the RPM™ seedlings was greater than that observed in bareroot seedlings, which experienced negligible diameter growth due to shoot dieback and resprouting. Increased initial height and diameter of RPM™ trees may be enough to overcome competition present in the floodplain. Additionally, a greater diameter growth rate may further increase the chance of survival. A small proportion of the swamp white oak RPM™ seedlings produced sound acorns in the first year. Production of acorns at young ages is a common feature of oaks grown using the RPM™ method. This feature is of interest to those managing for wildlife because it shortens the time between stand initiation and the production of hard mast.

These data represent the results after 1 year. We will continue to monitor seedling survival and growth to determine if stock type, species, soil mounding, or a cover crop improves oak regeneration as the competition intensifies and the sites experience flooding. The results of this research will lead to recommended methods of oak regeneration that can be used to

1) diversify native forests that develop on abandoned cropfields by supplemental planting of oak and other nut producing trees;
2) incorporate oaks and nut trees in agroforestry operations; and
3) sustain oaks in private land forests, greentree reservoirs, conservation areas, and wildlife refuges.

Humans always have and will continue to influence floodplain environments. Our hope is that by learning about processes of hard-mast reforestation, these habitats can be managed in more productive and valuable ways.

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THE DEVELOPMENT OF OAK REPRODUCTION FOLLOWING SOIL SCARIFICATION—IMPLICATIONS FOR RIPARIAN FOREST MANAGEMENT

John M. Lhotka and James J. Zaczek[1]

ABSTRACT.—With the current emphasis and interest in riparian forest management, it is necessary to develop management strategies that enhance and regenerate bottomland hardwoods in these biologically important areas. However, the regeneration of bottomland oaks has been problematic across much of the eastern United States. Two ongoing studies presented in this paper suggest that soil scarification, in the presence of abundant acorns, can increase the initial establishment of oak. One study assesses the effects of disk scarification on first year seedling establishment in a mixed-oak bottomland forest. The second study was conducted on an upland site within a fenced shelterwood and assesses the effects of bulldozer scarification on the development of seedlings 5 years after treatment. In both studies, the initial density of oak seedlings was greater and density and height of competitive tree species was reduced in the scarified areas than in the controls. Furthermore, the upland study showed that the benefits of scarification could be carried through year 5. From these studies, management recommendations have been developed and the implications of these recommendations for riparian management are presented. Finally, these studies suggest that soil scarification may be a useful tool for augmenting oak seedling reproduction in poorly regenerating riparian forests.

Riparian forests are some of the richest and most biologically diverse areas in the continental United States. They are home to a wide variety of plant and animal species, provide flood, erosion, sediment control, and function in nutrient cycling. In addition, these areas contain some of the most productive forest land in the United States and provide the forest products industry with several of the most economically important tree species (Kellison and Young 1997).

The vast complexity of site-species interactions in riparian forests makes the management of these forests difficult (Hodges 1997, Meadows and Stanturf 1997). As a result, the regeneration of many of these forests can be problematic. Likewise, past exploitive management practices employed in many of these areas have resulted in undesirable stand conditions (Meadows and Hodges 1997). Because of the diversity of benefits these systems provide, restoration or maintenance of riparian ecosystem health is of great importance.

To achieve a favorable species composition and structure in bottomland stands, the shelterwood system followed by appropriate intermediate stand treatments has been recommended to enhance seedling reproduction (Hodges and Janzen 1987, Loftis 1990, Meadows and Stanturf 1997). Treatments involving stand disturbance such as fire or soil scarification have been suggested as supplemental treatments to increase the establishment of oak reproduction (Scholtz 1959, Crow 1988, Barry and Nix 1993).

The principle behind soil scarification is that it improves germination and seedling establishment by incorporating acorns into the upper soil horizon(s). Studies have suggested that buried acorns are less susceptible to predation and have higher germination rates than acorns located on the soil surface (Griffen 1971, Janzen 1971, Auchmoody and others 1994, Nilsson and others 1996, Guo and others 1999). Soil scarification may also be useful in controlling competing understory vegetation, which may play a role in seedling development.

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Soil scarification has been tested, with favorable results, in both upland and bottomland oak forests under a variety of stand conditions and involving a diversity of species (Scholtz 1959, Zaczek and others 1997, Lhotka and Zaczek 2002). These studies suggest that soil scarification can enhance the initial establishment of oak seedlings in riparian forests. This paper highlights the effects of two recent scarification studies on the development of oak reproduction and their implications for riparian forest management.

ONGOING STUDIES

A soil scarification treatment was applied in 1993 on a poorly regenerating upland fenced shelterwood dominated by red maple seedlings and relatively few small oaks in central Pennsylvania. Results suggest that soil scarification in the presence of abundant viable acorns can increase the initial establishment and competitive position of oak and that the advantage is apparent through year 5 (Zaczek and others 1997). This upland study used a 78-hp crawler tractor with a 1.8-m wide brushrake to perform the scarification in autumn after mast dissemination.

One year after scarification, 32,618 oak seedlings ha^{-1} (78 percent of total seedlings) were found in the scarified plots while only 9,435 oak seedlings ha^{-1} were located in the control. The shelterwood overstory and fence was removed at year 3. Five years after treatment, a greater number of oaks (39,432 seedlings ha^{-1}) were still found in the scarified plots when compared to the control (12,971 seedlings ha^{-1}). Additionally, scarified areas had more oaks that tended to be in larger size classes compared to other species including red maple.

A second scarification study was conducted in the fall of 1999 mixed-oak bottomland forest in southern Illinois (Lhotka and Zaczek 2002). Primary oak species in the overstory were cherrybark oak (*Quercus pagoda* Raf.) and post oak (*Quercus stellata* Wangenh.). The understory had scant tree regeneration and was dominated by poison ivy (*Toxicodendron radicans* L.). A 42-hp wheeled tractor and 2.44-m wide field disk was used to complete the soil scarification treatment. One year after scarification, scarified plots had a greater density of oak seedlings (7,243 seedlings ha^{-1}, 42 percent of total seedlings) than did control plots (453 seedlings ha^{-1}, 9 percent of total). At the same time, poison ivy cover in the scarified plots (7 percent) was reduced compared to the control (35 percent). Results suggest the use of disk scarification has the potential of increasing the establishment and competitive position of desired oak species in riparian forests.

MANAGEMENT IMPLICATIONS

While soil scarification can increase the initial establishment of desired oak species, operational and biological influences may control the success of an operation. First, abundant viable acorns must be present at the time of scarification, for without the necessary seed, the success of a soil scarification operation can be severely limited. Extremely wet or dry conditions at the time of treatment may also influence the success of the operation. Extremely dry conditions may cause acorns to desiccate and loose their ability to germinate (Olson and Boyce 1971) and the depth of penetration by a disk in dry soils is limited. On the other hand, wet conditions may cause operational problems especially in riparian areas. With the soil excessively wet, scarification should not be implemented as the operation because it may compact soils and may bury acorns too deeply resulting in decreased seedling emergence.

Another important condition influencing a scarification operation is the presence of a dense midstory and understory. Stand density may limit the amount of space that machinery has to maneuver. The small-wheeled tractor with disk, small crawler tractor with brushrake or Salmon blade, and a modified drag-chain scarifier pulled by a small crawler tractor are three methods that may be used for scarification in partial harvests (Karsky 1993, Zaczek and others 1997, Lhotka and Zaczek 2002). These systems have the size and maneuverability to operate in a partially harvested stand without damaging residual trees, but still have sufficient power to complete the operation (Karsky 1993).

The disking and drag-chain method may be preferred in open stands, but the mobility of the equipment may be severely limited in dense stands or stands containing large amounts of downed slash. Therefore, it may be necessary to thin and/or remove slash prior to implementing the disk or drag-chain method. In contrast, the bulldozer/brushrake method has the ability to operate in dense and recently harvested stands, while still providing scarification benefits (Zaczek and others 1997, Zaczek in press). Damage to residual crop trees is possible so careful operation of equipment is necessary. For these reasons, it is important to consider stand conditions prior to planning a scarification operation.

Precision application of disturbance is another benefit that scarification can provide. Other oak regeneration enhancement treatments such as prescribed fire may be difficult or impractical to apply precisely in small or irregular-shaped patches and exclusion from sensitive areas may be problematic. With the ability to pinpoint treated areas, a manager could scarify poorly regenerating areas, while avoiding streamside buffer zones, temporal ponds, or other unique habitats. This flexibility makes the treatment useful with the complex variability in site conditions that occurs in riparian forests (Hodges 1997).

Soil scarification may enhance the establishment phase of a regeneration system, but this silvicultural treatment alone may not create appropriate growth conditions for the development of large, vigorous seedlings. Without the necessary cultural conditions in place, the continued growth and survival of seedlings produced after scarification may be limited. Therefore, it is recommended that soil scarification operations be implemented into a silvicultural system, such as a shelterwood, that provides the appropriate environmental conditions for the development of vigorous advance reproduction of oak.

CONCLUSION

Shallow soil scarification has been shown to enhance the density and competitive position of oak reproduction when applied during the fall after mast dissemination. It is cautioned that scarification should not be used during extremely wet conditions, on steep highly erodable slopes, or in sensitive habitats. Finally, soil scarification should be conducted along with a cutting prescription that promotes the development of advanced oak reproduction for without appropriate understory conditions seedling survival and growth may be limited.

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ADAPTATION OF FOUR AMORPHA SHRUBS TO FOUR LIGHT LEVELS

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Native legumes can play an important role in natural ecosystems and in tree plantings as a source of nitrogen through their symbiosis with rhizobial bacteria. The genus *Amorpha* of the subfamily Papilionoideae within the Fabaceae contains 20 to 25 shrubby species native to North America (Wilbur 1975). Several species are documented as nodulated by rhizobial bacteria (Allen and Allen 1981, Navarrete-Tindall 1998).

The most common *Amorpha* species, false wild indigo (*Amorpha fruticosa* L.) is found throughout most of the continental United States. It forms multiple stems reaching heights of 5 to 7 meters and has been used in windbreaks and for wetland restoration. The closely related smooth false indigo (*Amorpha nitens* Boyton), is documented from Kentucky and Arkansas and listed as an endangered species in Illinois and Georgia. Both false and smooth wild indigo can be found in canopy gaps following disturbance and in the understory of riparian and mesic hardwood forests (Taft 1994).

The *Amorpha* genus also includes smaller shrub species found in prairies including leadplant (*A. canescens* Pursh) and dwarf false indigo (*A. nana* Nutt). The objective of our studies was to evaluate the effect of four light levels on growth, development, and rhizobial nodulation of four Amorpha shrubs grown in pots in the greenhouse and under natural conditions.

In the experiment using natural conditions, we established two to three seedlings of false wild indigo, smooth wild indigo, leadplant, or dwarf wild indigo in 2 gallon white plastic pots filled with Metro-mix®-380. Previously, we collected seed of smooth wild indigo from plants found in two populations in Southern Illinois. Seed of other *Amorpha* species were purchased from a commercial nursery in Minnesota. We placed six pots of each species in each of nine shade structures located at the Horticulture and Agroforestry Research Center in New Franklin, MO.

Shade structures (5-m wide x 15-m long x 2.5-m high) in the University of Missour Center for Agroforestry outdoor shade laboratory were covered with shade cloth to maintain light levels of 20, 45, and 100 percent full sunlight. The experiment was established using a randomized block design with three replicates of each light treatment. The experiment ran from March to November for three consecutive years. We harvested aboveground stems each fall to determine oven-dried stem biomass. Stems with leaves were also harvested in early fall and late spring and analyzed for total plant nitrogen. We used the Combustion Analysis Technique (LECO), AOAC Official Method 990.03, to determine percent plant nitrogen.

In the greenhouse experiment, we established seedlings of false wild indigo, smooth wild indigo, and leadplant in D40 Deepots® filled with sterile vermiculite. Two pots of each species were placed in each of 18 shade frames set on benches in the greenhouses at the Horticulture and Agroforestry Research Center in New Franklin, MO. Light inside the greenhouse was approximately 50 percent of full sunlight and was supplemented with high-pressure sodium lamps to maintain a 16 hr photoperiod. Six of the eighteen shade frames were covered with shade cloth to reduce light levels by an additional 0, 55, or 80 percent.

We established the greenhouse experiment using a randomized block design with three

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replications of three light treatments with and without rhizobial inoculation. We inoculated half of the plants 1 and 3 weeks after establishment with a rhizobial mixture that included bacteria isolated from smooth wild indigo. All plants were watered as needed with sterile deionized water in addition to 50 ml Broughton and Dilworth solution biweekly (Somasegaran and Hoben 1994). We also added a potassium nitrate solution (6 mg N per plant) during the first and second week of the experiment. The greenhouse experiment ran for 4 months in two consecutive years.

At the conclusion of each experiment, we removed each plant from the vermiculite under water, measured stem and root length, and determined number of root nodules. Plants were oven dried to determine plant biomass and then ground to determine percent nitrogen using the LECO procedure.

Table 1.—*Stem biomass from sprouts on 2-year-old plants of four native* Amorpha *shrubs grown in outdoor shadehouses under three light levels*

Amorpha species	Light (% of full sunlight) 100 %	45 %	20 %
	- g -	- g -	- g -
False wild indigo	45 b	64 a	68 a
Smooth wild indigo	15 a	28 a	34 a
Lead plant	1.5 a	3.4 a	2.3 a
Dwarf wild indigo	2.6 a	5.8 a	5.4 a

Means within rows followed by different letters are significantly different ($p \leq 0.05$).

Table 2.—*Seasonal aboveground plant nitrogen in stems of four native* Amorpha *shrubs grown in outdoor shadehouses under three light levels*

Amorpha species	Harvest time	Light (% of full sunlight) 100 %	45 %	20 %
		- % -	- % -	- % -
False wild	Fall	2.3 a	2.2 a	2.0 a
Smooth wild	Fall	2.3 a	2.1 b	2.0 b
Leadplant	Fall	1.8 a	1.6 a	1.6 a
Dwarf	Fall	1.5 a	1.6 a	1.8 a
False wild	Spring	3.9 a	3.6 a	3.1 a
Smooth wild	Spring	3.5 a	3.1 b	2.8 c
Leadplant	Spring	2.8 a	2.4 a	2.6 a

Means within rows followed by different letters are significantly different ($p \leq 0.05$).

In the outdoor shade laboratory, we found biomass on sprout growth of 2-year-old false wild indigo plants was less when grown under full sunlight than when grown at under 45 and 20 percent of full sunlight (table 1). Although not statistically significant, the same trend was found for smooth wild indigo, leadplant, and dwarf wild indigo.

Of the four species, only smooth wild indigo showed decreasing aboveground plant nitrogen in response to decreasing light both in the fall and in the spring (table 2). When averaged over season and light intensity, false wild indigo had the highest plant nitrogen content (2.9 percent) followed by smooth false indigo (2.6 percent), then leadplant (2.1 percent), and, finally, dwarf indigo (1.6 percent). Plant nitrogen in the spring was nearly double that of plant nitrogen in the fall. One reason for the large differences may be that a 5- to 6-month-old stem was included with leaves in the fall samples while spring samples only had a 3- to 4-month-old stem with younger leaves.

In the greenhouse studies, we found a highly significant interaction among the three *Amorpha* species, with and without rhizobial inoculation, and three light levels (table 3). Although non-inoculated seedlings showed small changes in response to light, inoculated seedlings of all three species showed a marked decrease in plant biomass with decreasing light. Except for plants grown at 10 percent of full sunlight, shoot and root biomass for plants inoculated with rhizobia was 3- to 6-fold greater than plant biomass for non-inoculated plants. Plants grown in the greenhouse under 10 percent of

Table 3.—*Biomass of 4-month-old plants under three light levels for three* Amorpha *shrubs with and without rhizobial inoculation*

Amorpha species	Rhizobia	Light (% full sunlight) 50 %	22 %	10 %
		- g -	- g -	- g -
False wild	With	2.3 a	2.5 a	0.5 b
Smooth	With	1.2 a	0.6 b	0.4 c
Lead plant	With	2.0 a	1.1 ab	0.2 b
False wild	W/out	0.4 a	0.4 a	0.2 b
Smooth	W/out	0.3 a	0.3 a	0.1 b
Lead plant	W/out	0.7 a	0.4 a	0.1 a

Means within rows followed by different letters are significantly different ($p \leq 0.05$).

full sunlight were distinctly etiolated, seldom upright, and had the fewest nodules if inoculated.

In general, there was a trend for nitrogen content to decrease slightly with decreasing light intensity 4-month-old plants with and without rhizobial inoculation (table 4). We found that the number of nodules decreased with decreasing light levels and that non-inoculated plants could be nodulated from air-borne rhizobia (data not included).

The average plant nitrogen concentration across all light levels was slightly higher (2.4 percent) for smooth wild indigo than for wild false indigo (2.2 percent) and leadplant (2.0 percent). The lower values for plant nitrogen in the greenhouse plants (table 4) compared to those in the outdoor shade tolerance laboratory (table 2), may be a consequence of including the root and nodules along with the stem and leaves for the greenhouse-grown seedlings.

Our results suggest that the riparian species, false wild indigo and smooth wild indigo, and the prairie species, leadplant and dwarf wild indigo, are tolerant of moderate shade and were readily nodulated by native rhizobial bacteria. All four *Amorpha* species should be evaluated further for possible inclusion in various agroforestry practices as woody nurse crops or in savanna restoration projects. Field trials are needed to test how effectively nitrogen is lost from these species and made available to other plants in managed ecosystems.

Table 4.—*Plant nitrogen concentration for three* Amorpha *shrubs grown for 4 months under three light levels with and without rhizobial inoculation*

Amorpha species	Rhizobia	Light (% of full sunlight)		
		50 %	22 %	10 %
		- % -	- % -	- % -
False wild	With	2.0 b	2.4 a	2.1 b
Smooth	With	2.7 a	2.3 b	2.3 b
Leadplant	With	2.1 a	2.2 a	2.1 a
	MEAN:	2.3	2.3	2.1
False wild	W/out	2.4 a	1.9 b	2.4 a
Smooth	W/out	2.7 a	2.5 ab	2.1 b
Lead plant	W/out	2.0 ab	2.3 a	1.8 b
	MEAN:	2.4	2.3	2.1

Means within rows followed by different letters are significantly different ($p \leq 0.05$).

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SOIL WATER NITRATE CONCENTRATIONS IN GIANT CANE AND FOREST RIPARIAN BUFFER ZONES

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ABSTRACT.—Soil water nitrate concentrations in giant cane and forest riparian buffer zones along Cypress Creek in southern Illinois were compared to determine if the riparian zones were sources or sinks for nitrogen in the rooting zone. Suction lysimeters were used to collect soil water samples from the lower rooting zone in each of the two vegetation types. The cane riparian zone had a significantly lower mean soil water nitrate-N concentration (0.45 mg L^{-1}) than the forest riparian buffer zone (4.29 mg L^{-1}). This was attributed to higher litter C/N ratios in the cane, which likely resulted in greater microbial immobilization of nitrogen within the rooting zone.

Forest and grass riparian buffer zones are being promoted throughout the United States and abroad to combat agricultural non-point source pollution. In Illinois, a leading agricultural state, nutrient and sediment inputs into streams are the States top water quality impairments. Excess nutrient export from the midwestern agricultural states, including Illinois, has been linked to the creation of hypoxic zones in the Gulf of Mexico (Alexander and others 2000).

Most of midwestern riparian zone research has focused on tile-drained agricultural systems in central Illinois and central Iowa (Osborne and Kovacic 1993, Schultz and others 1995, Lee and others 2000). However, the majority of agricultural land in the Midwest, including southern Illinois, is not tile-drained. Therefore, we are investigating the ability of native Illinois riparian cover types (cane and forest) to serve as sinks for nitrogen within the rooting zone of a non tile-drained system.

Historically, giant cane was an important component of the pre-European settlement landscape in southern Illinois (Flower 1822). There is significant interest among the state resource agencies responsible for riparian restoration in returning cane to the local landscape. There has been no research conducted to evaluate the nutrient attenuation capabilities of giant cane. The data collected from this research will supplement vegetation restoration and establishment guidelines for southern Illinois riparian zones.

MATERIALS AND METHODS

Study Location

The riparian research area was located in the Cypress Creek Watershed. Cypress Creek is a tributary of the Cache River in southern Illinois, and is one of the Illinois Department of Natural Resources (IDNR's) pilot watersheds (fig. 1). Field plots in two vegetation types (cane and forest) were established and monitored along Cypress Creek on a privately owned farm, Section 3, Township 13 South, Range 3 East (USGS 7.5 Topographic Map). The cane and forested plots were estimated to be 20-30 years old. The contributing agricultural field that drained into the buffers was 0.26 ha in size and had an average slope of 1 percent. The soil within the study area is classified as Haymond silt loam (Typic Udifluvent) and the underlying geology of the area is Ste. Genevieve (or Levias) and Warsaw limestones.

A no-till rotation between corn (*Zea mays* L.) and soybeans (*Glycine max* (L.) Merr.) had been practiced, with fertilizers added every other year (each corn year). Dry fertilizers, 0-0-60 and 18-

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46-0, were added to the soil surface at a rate of 223 kg ha^{-1}. Ammonia, at a rate of 167 kg ha^{-1} was knifed into the soil before planting. In the early spring, before planting, dense winter annuals covered the agricultural field, reducing the risk of erosion.

Field Methods

Water samples were collected between November 2000 and October 2001 at the study area. Soil suction or vacuum lysimeters were installed within the rooting zones of the vegetated buffer zones to sample soil water. The lysimeters were installed to a depth of 0.2 m in the cane buffer and 0.6 m deep in the forested buffer. The two depths represented the rooting depths of the two cover types. Within the two vegetation plots, 12 lysimeters were installed in three rows of four perpendicular to the field edge to try to account for the variability in soil water chemistry. The lysimeters were installed at 3.3 m intervals starting from the field edge. Row locations were randomly selected within the plot boundaries.

Soil lysimeters were composed of a 5 cm PVC tube, a porous ceramic cup, and a rubber cap. Lysimeters were installed on 2000 June 7. Sixty pounds of vacuum pressure was applied to the lysimeters using a hand pump to create pressure to extract water from the soil micropores. For the first 5 months after installation, the lysimeters were not sampled but flushed every 2 weeks to allow microbial activity to return to normal rates after the disturbance. Starting in November 2000, samples from the lysimeters were collected bi-monthly, and after significant storm events.

At the beginning of the study, soil water samples were analyzed for ammonium, nitrate, and orthophosphate concentrations. After 3 months, samples were only analyzed for nitrate because ammonium and orthophosphate concentrations were always at or below the detection limit (0.05 mg L^{-1}). However, samples were randomly checked for ammonium and orthophosphate throughout the study period and none were detected. During the mid summer months, the soils became too dry for most of the lysimeters to extract water. During this period, measurable samples could be collected from only 1 or 2 lysimeters. Droughty conditions persisted at the site through September. In the wetter months, nearly all lysimeters accumulated sufficient water for sampling.

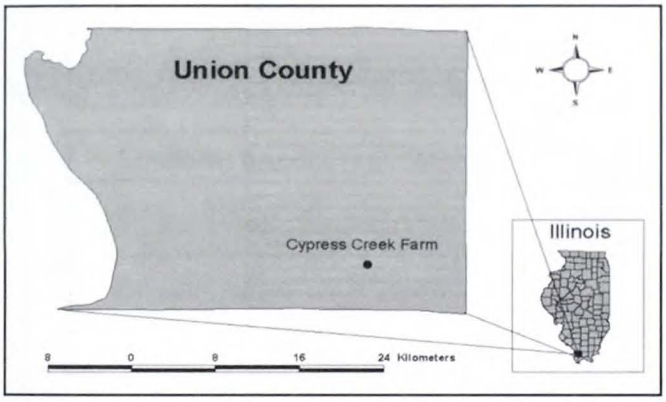

Figure 1.—*Study location along Cypress Creek in Union County, Illinois.*

The two vegetation communities were characterized by estimating the biomass and stems per acre for each site. To estimate cane biomass, 30 stems were randomly selected to represent the population of stems within the buffer. The leaves and the culms were separated and oven-dried to estimate their individual biomass. To estimate the number of stems per acre, a stratified random sample was used to locate 10 m^2 plots. Within each plot, each stem was counted and 10 random stems were measured for their diameter. In the forested site, all trees were identified and measured at diameter breast height (DBH). With published biomass equations, DBH was used to estimate the total aboveground biomass for each species in the forested cover type (Ter-Mikaelian and Korzukhin 1997).

Soil within the cane and forested buffers were analyzed for physical and chemical characteristics. The soil texture was analyzed using a hydrometer method outlined by Black and others (1965). Ten soil cores were collected at the soil surface, measured for volume and weighed. The cores were then oven dried and re-weighed to calculate the bulk density (Black and others 1965). The infiltration rates were also measured using ring infiltrometers (Black and others 1965). Five samples were taken at each site and the geometric mean was used to account for the variability in sampling locations.

Laboratory Methods

Soil water nitrate concentrations were analyzed using a Dionex 4000i ion chromatograph (Dionex Corporation, Smyrna, GA). Soil water ammonium and orthophosphate were analyzed with a Hach 4000v spectrophotometer via a nesslerization and absorbic acid method, respectively (Greenberg and others 1992). Total carbon to total nitrogen (C/N) ratios of the litter

and the mineral soil were also determined for the two vegetation communities. First, the 4.5 m cores were collected with the help of Union County Natural Resources Conservation Service (NRCS) personnel. An NRCS soil scientist classified the soils.

The different horizons of the soil cores were individually analyzed for their C/N ratio. The soil was prepared for C/N determination by air drying, then sieving through a 2 mm (9 mesh) screen. Six mg of fine, dry soil was placed in 5 mm X 9 mm tin capsules along with an even scoop of vanadium pentoxide. Eighty-nine capsules containing mineral soil and litter samples were shipped to Fernow Experimental Forest Station in Parsons, WV, for total C and N analysis by combustion using an organic elemental analyzer (Baccanti and others 1993). The resulting total C and N values were reported on a percentage basis, mg of C or N per mg of soil. These total C and N percentages were converted to kg ha^{-1} of C and N by multiplying the percentages by the bulk density (g cm^{-3}) and the depth of the surface mineral soils, 15 cm.

Statistical Methods

Soil water nitrate-N was analyzed with SAS for Windows V8 (SAS Institute, Cary, NC). A mixed model with repeated measures procedure was used to analyze soil water nitrate-N differences between cover types and within cover types at varying distances. The LS MEANS mean separation procedure was used to identify differences between the sites. A T-test with unequal variances was used to analyze differences between the C/N ratios in soil cores.

RESULTS AND DISCUSSION

The sand, silt, and clay content within both the cane and forested riparian buffer zones were 10, 70, and 20 percent, respectively. The bulk density of the cane site was 1.08 g cm^{-3} and the forested site was 1.19 g cm^{-3}. The infiltration rate of the cane site was 37.93 mm hr^{-1} and the forest was 26.78 mm hr^{-1}.

Soil water within the forested riparian buffer zone had a significantly (alpha = 0.05) higher nitrate-N concentration (4.29 mg L^{-1}) than within the cane buffer zone (0.45 mg L^{-1}) (fig. 2). It

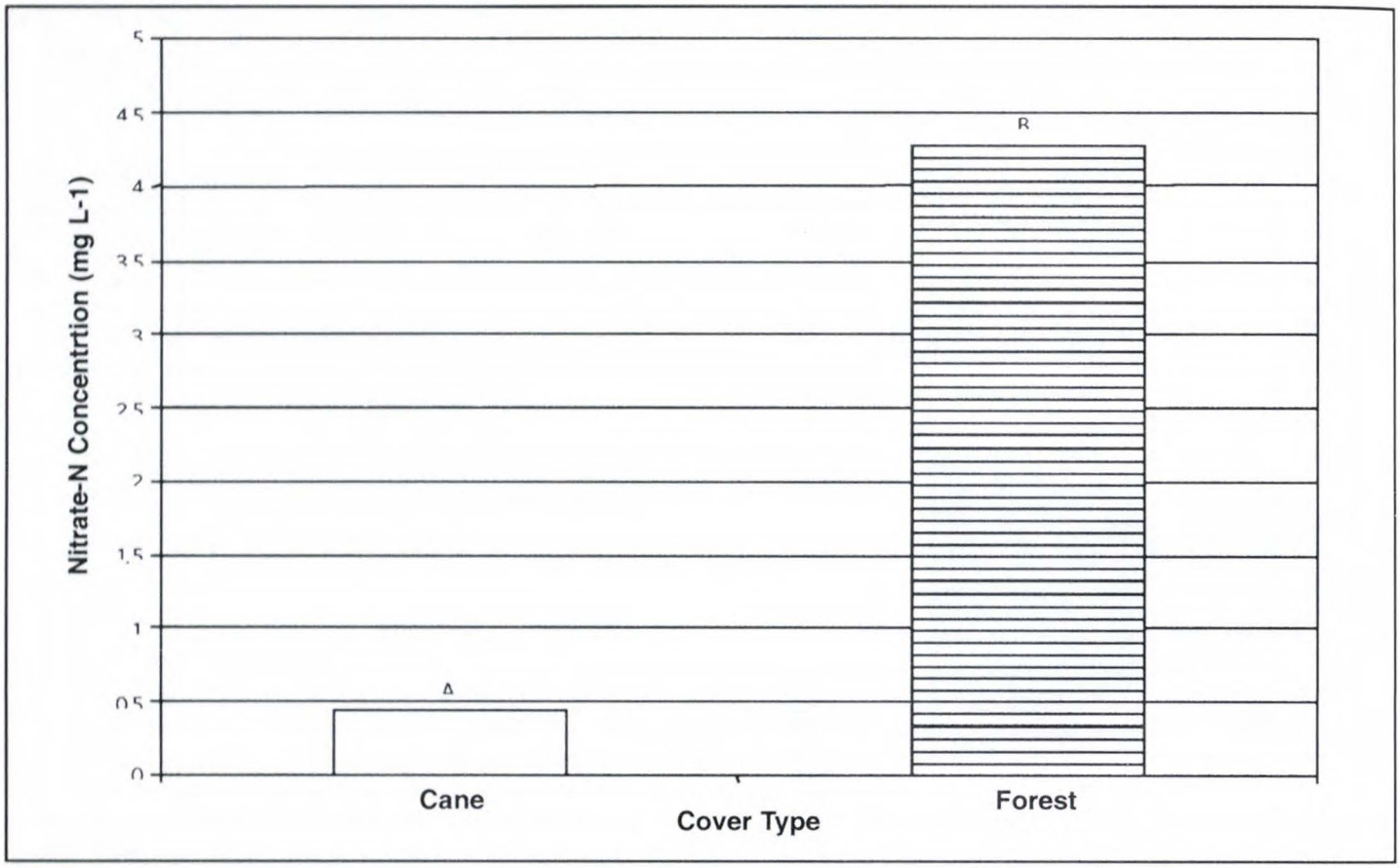

Figure 2.—*Mean nitrate-N concentrations in soil water beneath giant cane and forested cover types.*

is assumed that nitrogen inputs from the agricultural area into the two vegetation plots were similar because the plots drained the same field and were adjacent to one another. It is also significant to note that our soil horizons did not contain any shallow confining layers or fragipans. Therefore, we assumed that the majority of soil water sampled in the riparian zones originated from vertical percolation of overland flow within the riparian areas and not lateral interflow from the agricultural field.

The lower average nitrate concentration in the cane could be explained by differences in root uptake rates and/or microbial nitrogen cycling rates between cane and forest. The cane could have greater uptake rates of nitrate due to an observed dense rooting network in the shallow soil horizons. While root biomass was not estimated for the two cover types, the aboveground biomass was calculated. The forest had significantly (alpha = 0.05) greater aboveground biomass (463,717 kg ha^{-1}) than the cane (36,321 kg ha^{-1}), which does not support our results assuming aboveground biomass is an adequate index to root biomass.

The soil nitrogen cycling characteristics of the two cover types was investigated by measuring the C/N ratios in the soil at rooting depth. At C/N ratios >30, net N immobilization generally occurs while at C/N ratios <20, net N mineralization occurs (Alexander 1977). This is due to the competition for N between nitrifying (autotrophic) and immobilizing (heterotropic) bacteria in the soil. It has been shown that heterotrophic bacteria can out compete nitrifiers for available ammonium (Riha and others 1986). Thus, at lower C/N ratios, the heterotrophic demand for ammonium is satisfied and more N is available for nitrifying bacteria, allowing for greater net nitrate production. The cane litter had a significantly greater mean C/N ratio (20.47) than the forest (17.12) (table 1), indicating a greater likelihood of net N immobilization rather than net mineralization. C/N ratios of litter have been found to be a strong indicator of the potential for nitrate leaching in forest soils in a broad survey of European forests (Gunderson and others 1998).

Increased N immobilization in the cane site may also be due to a more uniform addition of labile carbon to the litter layer via year-round leaf-fall. During forest leaf-fall in mid- to late-October, a flush of carbon would be added to the system, but throughout the remainder of the year the litter would likely be left containing more recalcitrant carbon forms (Paul and Clark 1996).

C/N ratios in mineral soil were higher in the forest soil (13.61) than the cane soils (10.96) (table 1), which is opposite of the relationship in the litter layer of the two sites. If carbon was in an available form in the mineral soil, one would expect more net nitrogen immobilization in the forest soils. However, tree leaves, bark, and stems contain substantial amounts of carbon that is in recalcitrant forms, such as lignin (Paul and Clark 1996), while grasses have fewer recalcitrant types of carbonaceous compounds within their tissues (Paul and Clark 1996). Therefore, the cane may still exhibit greater net N immobilization than the forested site, because its carbon is found in more available forms that are rapidly assimilated. The rapid assimilation of the carbon could result in the lower observed C/N ratios within the mineral soil of the cane.

Microbial denitrification within the soils is another possible source of nitrate loss within the rooting zone. However, at our study site, denitrification was assumed to be negligible because of the presence of a relatively well-drained riparian soil, and a water table that remained well below the rooting zone throughout the study.

Table 1.—*Carbon to nitrogen ratios (C/N) of mineral soil and litter in forest and giant cane riparian buffer zones*

Site	Location of core (m)	Mineral soil C/N	Litter C/N	Total N content of surface soil kg ha^{-1}
Cane	Field edge	12.48	21.74	1,177
Cane	3.3	10.50	21.28	1,308
Cane	6.6	10.75	21.26	2,452
Cane	10.0	10.11	17.60	2,779
Forest	Field edge	14.38	16.85	1,805
Forest	3.3	14.92	17.15	3,203
Forest	6.6.	15.19	16.87	2,690
Forest	10.0	9.97	17.61	2,655

CONCLUSIONS

Giant cane appeared to be a greater sink for nitrogen than deciduous forest. High litter C/N ratios of cane likely contributed to the low soil water nitrate-N concentrations by stimulating microbial nitrogen immobilization. The year-round inputs of fresh litter by cane supplied significant amounts of labile carbon to riparian soils, likely resulting in substantial microbial nitrogen immobilization. Giant cane's ability to serve as a sink for nitrogen in the rooting zone provides further evidence why it should be included in a multi-species riparian zone restoration design.

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EFFECT OF VEGETATION STRUCTURE ON BREEDING TERRITORY SELECTION BY RED-WINGED BLACKBIRDS IN A FLOODPLAIN FOREST RESTORATION PROJECT

Maria A. Furey, Dirk E. Burhans, Hong He, Michael A. Gold, and Bruce E. Cutter[1]

Our research investigates the role of vegetation structure in the selection of breeding territories by red-winged blackbirds (*Agelaius phoeniceus*) in two floodplain oak-restoration sites. Perches are used extensively by red-winged blackbirds in territorial display during the spring (Yasukawa and Searcy 1995). We hypothesized that breeding territory size may be determined by perch availability, and that breeding territories with few perches would require a larger area for male red-winged blackbirds to defend against other males. We predicted that breeding territories exhibiting high perch densities would be smaller than breeding territories with low perch density. Additionally, we were able to test the importance of perch availability in a floodplain restoration setting in which oaks were planted in varying patterns, and in which a cover crop in one plot type reduced availability of natural perches.

The larger study objectives are to determine:

1) how the availability of perches determines the size of breeding territories in Red-winged blackbirds,
2) how the availability of perches determines the spatial location of breeding territories within a plot type,
3) how occupied areas compare in vegetation composition and structure from unoccupied areas within the plots.

In this paper we present preliminary results concerning the first objective.

Our research is being conducted in the central region of Missouri on two sites located within the Missouri River floodplain. Plowboy Bend Conservation Area (PBB) is a row crop agriculture/floodplain ecosystem. The research area resides west of the Missouri River's main channel within a levee-protected floodplain. Smoky Waters Conservation Area (SW) resides between the main channel of the Missouri and the Osage River, a major tributary of the Missouri River. Smoky Waters floodplain has not been protected by a levee since the levee was breached in the 1993 and 1995 great floods.

Both study areas contain three 40-acre research plots of distinct vegetation treatments established in 1999 for a separate research project examining hard mast (oak acorn) restoration (Shaw and others 2003). "Redtop" plots consist of a random block design featuring oak saplings of swamp white oak (*Quercus bicolor* Willd.) and pin oak (*Quercus palustris* Muenchh.) of varying greenhouse methods, seedbed treatments, and a uniform ground cover of grass (*Agrostis gigantea* Roth). The redtop ground cover has reduced infiltration of other herbaceous vegetation used as perches by red-winged blackbirds. "No Redtop" plots contain the same random block design of treatments without a seeded ground cover. "Control" plots do not have any of the vegetation treatments found in Redtop or No Redtop plots.

We identified breeding territories by monitoring male red-winged blackbirds performing mating display and territory defense behaviors. Territory delineation was accomplished using consecutive flushing (Wiens 1969), a technique in which the bird is approached and followed until he alights on perches defining his territory. Perch data were collected using visual observation and a height pole. Perch density (vegetation exceeding 1.5 m heights) was measured using belt transects placed in two cardinal directions within the territory. We analyzed effect of site and plot type and stem density on territory area by analysis of covariance.

Mean breeding territories at Plowboy Bend were 2,417 ± 354 m^2 (SE; n = 20) and at Smoky

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Waters were 1,743 ± 153 m² (n = 36). There was no effect of stem density, site, and plot type on size (area) of red-winged blackbird breeding territories (table 1; overall $F_{4,48}$ = 1.60, P = 0.19; site: $F_{1,48}$ = 3.08, P = 0.09; plot: $F_{2,48}$ = 1.14, P = 0.33; stem density: $F_{1,48}$ = 0.05, P = 0.82).

Mean perch densities within breeding territories were 1.6 ± 0.3 m² (n = 18) at Plowboy Bend and 3.0 ± 0.6 m² (n = 35) at Smoky Waters (table 2). We did not find an effect of perch density on size (area) of red-winged blackbird breeding territories. We also did not find an effect of plot type on breeding territory size; however, results suggested the possibility of a difference in territory area between the two sites, and perch density may vary between the two sites as well.

Future work during the 2002 field season will increase sample size to further test for these effects. In addition, we will test the effects of horizontal and vegetation structure on breeding territory area, and determine the relationships of these aspects to breeding territory area.

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Table 1.—*Mean ± standard error (n) of breeding territory area within two conservation areas by plot type*

	Mean territory area and SE (m²)	
Plot Type	Plowboy Bend	Smoky Waters
Control	3,731 ± 3,557 (2)	2,156 ± 246 (8)
No redtop	2,260 ± 284 (15)	1,638 ± 277 (8)
Redtop	2,326 ± 527 (3)	1,621 ± 230 (20)

Table 2.—*Mean ± standard error (n) of perch density within breeding territories at two conservation areas by plot type*

	Perch density (# stems/m²)	
Plot type	Plowboy Bend	Smoky Waters
Control	0.98 ± 0.72 (2)	1.16 ± 0.40 (8)
No redtop	1.49 ± 0.36 (13)	3.46 ± 1.50 (7)
Redtop	2.29 ± 0.21 (3)	3.60 ± 0.79 (20)

EFFECT OF SITE TREATMENTS ON SOIL TEMPERATURE AND MOISTURE AND OAK AND PINE GROWTH AND NUTRIENT CONCENTRATIONS

Felix Ponder, Jr.[1]

ABSTRACT.—Five years after planting, measurements of soil moisture and temperature, leaf nutrient concentrations and growth, were compared for plots of northern red oak, white oak, and shortleaf pine for treatment combinations that included two levels each of harvesting intensity (organic matter removal), site disturbance (soil compaction), and weed control (control of the understory). There were significant interactions between organic matter removal and soil compaction for height, height growth, and d.b.h. of northern red oak, and for the height of white oak, but not for shortleaf pine. Controlling the understory contributed to a marked increase in height, height growth, and d.b.h. of northern red oak and white oak, but only the d.b.h. of shortleaf pine. Mean soil moisture and temperatures were significantly higher for treatment combinations without an understory. Also, with some exceptions, leaf nutrient concentrations were higher for treatment combinations without an understory compared to treatment combinations with an understory. The two-way and three-way interactions for the effect of treatments on variables increased the difficulty of sorting out dominant influences among main effects. However, at age 5, interactions involving understory vegetation seemed to impact soil moisture and temperature, tree nutrition, and growth as much or more than organic matter removal or soil compaction on this site.

The effects of soil compaction and organic matter removal (above ground biomass) on soil moisture and temperature which alter tree growth are not well understood; even less is known about how soil compaction and organic matter removal affect nutrient uptake. Forested ecosystems contain large amounts of nutrients in woody biomass that may exist either as standing material, on the soil surface, or within the soil profile. Whole-tree harvesting and debris removal remove considerable amounts of nutrients (Ponder and Mikkelson 1995); and the disturbance caused by the process may sometime later, if not immediately, affect nutrients left on the site by increasing soil erosion, mineralization, and leaching.

Wells and Jorgensen (1979) concluded that because soil nutrient supply and productivity in forests change relatively slowly, biomass-harvesting practices could be used for several rotations without serious risk of decline in soil productivity. However, an increase in the amount of biomass harvested can be expected to increase soluble nutrient losses and increase transport of particulate matter. Also, increasing the amount of biomass removal reduces the quantity of organic residue that would ordinarily decompose and release nutrients. If forest floor temperature and moisture are increased by biomass removals, there could be a nutrient flush from accelerated forest floor decomposition.

Soil compaction can contribute to poor root health and reduce root response time to localized nutrient concentrations (Shierlaw and Alston 1984, Chaudhary and Prihar 1974). Soil compaction has been shown to cause reductions

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in soil volume, soil porosity, aeration, water infiltration, and saturated hydraulic conductivity (Graecen and Sands 1980), or increase bulk density, which limits root growth (Daddow and Warrington 1983).

Castillo and others (1982) reported that mechanical stress to soil while pea seedlings (*Pisum sativum* L. cv. Alaska) were actively growing, decreased the uptake of Ca, Mg, and Mn, but the uptake of Cu was increased. These authors also observed differences in nutrient uptake associated with differences in soil water potential. Compressing soil around growing roots during field operations may reduce root length and nutrient uptake to a greater extent than will occur when plants are grown in initially more compacted soil.

The USDA Forest Service initiated research to study the impact of soil compaction and organic matter removal on the growth and development of regenerating vegetation and long-term soil productivity (LTSP) of forested lands in the United States (Powers and others 1989, 1990). Similar studies have been established in Ontario and British Columbia, Canada. The study has two major objectives. The first is to determine how changes in soil porosity and organic matter affect soil processes controlling productivity and sustainability and secondly, to compare results from similar replicated studies among major forest types. For this report, only the low and high treatments were used. This report compares measurements of soil moisture and temperature, as well as growth and leaf nutrient concentrations of planted northern red oak, white oak, and shortleaf pine after five growing seasons in treatments that included two levels each of organic matter removal, soil compaction, and weed control at the Missouri site.

MATERIALS AND METHODS

Study Site

The study is located at the Carr Creek State Forest in Shannon County, MO. Shannon County is located in the southeastern Missouri Ozarks. Mean annual precipitation in the area is 112 cm and mean annual temperature is 13.3°C (Barnton 1993). The study site is located on the upper northeastern-facing side slopes (20 to 28 percent slopes) of two parallel ridges. The weathering of the Ordovician and Cambrian dolomite has resulted in a deep mantle of cherty residuum (Gott 1975). Soils derived from this residuum are primarily of the Clarksville series (loamy-skeletal, mixed, mesic, Typic Paleudults). Prior to harvest, the site had a well-stocked, mature, second-growth oak-hickory forest. The site index for 50-year-old black oak (*Quercus velutina* Lam.) ranged from 22.5 to 24.3 m (74 to 80 feet) (Hahn 1991).

Experimental Design

The LTSP study includes nine treatments derived from combinations of three levels each of organic matter removal and soil compaction. The three levels of organic matter removal included:

1) merchantable boles removed (boles only, OM_0),
2) all living vegetation removed (whole tree, OM_1), and
3) all living vegetation plus forest floor removed, exposing mineral soil (whole tree + forest floor, OM_2).

Merchantable boles included trees with diameters at breast height (d.h.b.) of 25 cm or larger. The three levels of compaction included:

1) no compaction (C_0),
2) moderate compaction (C_1), and
3) severe compaction (C_2).

The targeted bulk density of severe compaction treatment was an increase of 30 percent more than the bulk density of the no compaction treatment. The moderate soil compaction treatment was intermediate between the severe compaction and no compaction treatments. Each of the nine treatment combinations was replicated three times in plots approximately 0.4 ha in size before trees were harvested.

Trees were harvested from February through May 1994. On plots designated as no compaction (C_0), all trees with a d.b.h. of 25 cm or larger (merchantable trees) on OM_0, OM_1, and OM_2 plots were directionally felled and removed with a skyline cable logging system. Merchantable trees on plots where the soil was to be compacted (OM_1 and OM_2), plot borders, and the area within the study boundary were directionally felled and removed with a skidder that traveled only on designated paths within the plots and in plot borders. After the removal of merchantable trees in OM_0 plots, the remaining trees were felled. This standing biomass included trees with d.b.h. less than 25 cm (unmerchantable trees), crowns from merchantable trees, standing dead and live snags, and stems of tree species and shrubs in the herbaceous layer with basal diameters greater than 2 cm or at least 25 cm tall. Trees were cut into lengths that permitted material to be either hand carried off plots or distributed over the plots, depending on the OM removal treatment.

Except for crowns, which were retained on the OM_0 plots, but not on OM_1 plots, the remaining biomass on OM_1 plots was treated the same as biomass on OM_0 plots.

On OM_2 harvested plots, all biomass remaining after harvest was removed. This included all understory vegetation plus the forest floor was raked away to the mineral soil. For OM plots requiring compaction, it was necessary to remove and replace these materials (except the leaf/litter layer) before and after compaction. Skidders and tractors were permitted on compacted plots, but not on plots that were not to be compacted. A 14-ton vibrating sheep-foot roller was used to compact soil. The roller made passes over the severe compaction treatment until there was no change in bulk density after roller passes.

For bulk density measurements, soil cores (30 cm in length x 9.2 cm in diameter) were extracted from each plot using a soil-coring device (Ponder and Alley 1997). The cores were divided into 10-cm depth increments, oven dried at 105°C, and weighed. Bulk density for each sample was calculated according to the method of the Soil Survey Staff (1984). Soil bulk density measurements were taken after the roller made one, three, five, and eight passes over plots that received the severe compaction treatment. Initial soil chemical properties of the 0 to 20 cm depth were: 5.33 percent OM, 3.29 percent total C, 0.11 percent total N, 16 $\mu g\ kg^{-1}$ P, 789 $\mu g\ kg^{-1}$ Ca, 61 $\mu g\ kg^{-1}$, and pH of 5.7.

In late spring of 1994, each of the 27 plots were planted to a mixture of 1-0 red oak (*Quercus rubra* L.), white oak (*Q. alba* L.), and shortleaf pine (*Pinus echinata* Mill.) in a 3:3:1 ratio, (three red oak seedlings plus three white oak seedlings to one shortleaf pine seedling), using hoedads. Seedlings were planted in rows at 2.5 m x 2.5 m apart. A 1-m radius area around each seedling was sprayed with a glyphosate and simazine mixture to control weeds for the first 2 years. Beginning in the third growing season, half of each plot was kept weed-free (U_-) to permit planted trees to grow freely. Weeds were not controlled (U_+) in the other half of the plot.

Seedling diameters at 2.5 cm above ground and heights were measured after planting and annually thereafter. Diameter at breast height (d.b.h.) was measured when trees reached 1.4 m tall or taller. Leaf samples were collected the fifth year of the study and analyzed for nutrient element concentrations. Soil moisture and temperature at 10, 20, and 30 cm deep were measured monthly during the growing season using Soiltest™ moisture cells. For this report, measurements for the 20 cm depth were selected without bias. All measurements, leaf sampling, preparation, and analyses were done according to standard procedures. Here, soil moisture and temperature measurements for 2 years at the 20 cm depth, and two levels each of organic matter removal and soil compaction, with and without understory control, and their impact on seedling growth after 5 years are reported.

Statistical Analyses

The experiment was analyzed as a split-plot design with three replications, with organic matter removal and compaction treatments as the main plots and the weed control treatments as the subplots. Analysis of variance procedures were conducted with the PROC GLM procedures in SAS (Version 6, SAS Institute, Cary, NC). All statistical tests were performed at the $\alpha = 0.05$ level of significance.

RESULTS AND DISCUSSION

Soil Temperature

Statistics for treatment effects on soil moisture and temperature for the fourth and fifth growing seasons are presented in table 1. The greatest difference in soil temperature between treatments was between U treatments. However, there was a significant interaction between OM and U. Soil temperatures were higher in the OM_2 treatment when the U was absent. The significance three-way interaction for year 4 between OM removal, C and U treatments showed that compacted plots tended to have higher temperatures than plots not compacted when the OM removal level was OM_0; and temperatures were always higher when the U was present compared to when it was absent.

The three-way interaction was not significant in year 5, but a two-way interaction between OM removal and U control was (table 1). Likely, considerably more heat was absorbed in U_- plots compared to U_+ plots because of the higher soil moisture content. In a regenerating clearcut such as this, most of the sunlight reaches the forest floor. Seasonal soil temperature differed little between OM removal and C treatments in year 4. However, soil temperature variations between treatment levels were much more apparent for each treatment throughout the measurement period for year 5 (data not presented).

Table 1.—*Mean soil moisture and temperature measurements and their p value if greater than 0.05 for significant interactions at the 20 cm depth, according to treatments for the fourth and fifth growing season in the Missouri LTSP study*

Treatment[1]	Soil moisture		Soil temperature	
	Year 4	Year 5	Year 4	Year 5
	-------- Percent------		---------- °C ----------	
Organic matter removal				
OM_0	28.33 a	23.23 a[2]	18.08 a	11.62 a
OM_2	25.81 a	20.84 a	18.19 a	14.88 b
Compaction				
C_0	29.29 a	25.29 a	17.89 a	11.88 a
C_2	24.81 b	19.30 b	18.38 a	14.21 a
Understory				
U_-	27.65 a	33.54 a	16.45 a	16.30 a
U_+	24.85 a	6.31 b	19.56 b	10.48 b
OM x C	NS	0.0035	0.0002	NS
OM x U	NS	NS	0.0243	0.0012
C x U	NS	NS	NS	NS
OM x C x U	NS	NS	0.0001	NS

[1]Organic matter removal = bole only (OM_0) and whole tree plus forest floor (OM_2). Soil compaction = none (C_0) and severe (C_2). Understory = understory absent (U_-) and understory present (U_+).

[2]Means followed by the same letter are not significantly different (NS) from each other ($P \geq 0.05$) within a column and treatment by Tukey's test.

The increase in soil temperature for OM_2 plots over OM_0 plots in year 5 may be attributed to exposure of the forest floor to sunlight and the lack of a forest floor litter layer in the OM_2 treatment to insulate the soil compared to OM_0 where the forest floor and litter was not removed. Carey and others (1981) observed that harvest residues created an insulating layer over the site. Messina and others (1997) reported that in addition to sunlight reaching the forest floor, microbial and root activity may also cause higher temperatures, especially following forest disturbance.

Soil Moisture

Soil moisture at the 20 cm depth differed significantly between compaction levels in year 4 (table 1), averaging almost 4 percent higher in the C_0 treatment than in the C_2 treatment. In year 5, there was a significant interaction between OM removal and soil C treatments which showed that soil moisture was highest for OM_0C_0, decreased more by the combination of OM_0C_2 than by OM_2C_2 or OM_2C_0. Also, in year 5, soil moisture differed between levels of U control. Soil in U_- treatments had 27 percent more moisture than soil in U_+ treatments. Undoubtedly, U made a major difference in soil moisture at the 20 cm depth on this site. Fifth year soil moisture data showed wider differences between treatment levels, and low, rather than high treatment levels, plus controlling the U produced higher soil moisture at the 20 cm depth.

The insulating effect of debris remaining after harvest on OM_0 plots reduced moisture loss from evaporation. Also, the degree of variation in soil moisture is also dependent on type and amount of vegetative cover, particularly in both clearcut hardwood and pine stands (Haines and Cleveland 1981).

Soil moisture was lower for treatment combinations in C_2 plots than for C_0 plots for both year 4 and for year 5. These results indicate that compaction reduced soil moisture at the 20 cm depth for this site. Soil bulk density at planting at the 20 cm depth was 1.49 g/cm^3 for C_0 plots compared to 1.88 g/cm^3 for C_2 plots (Ponder 1997).

Understory did not affect soil moisture in year 4 but did in year 5 (table 1). The lack of effect in year 4 may be due to the greater than usual rainfall for that year and over the measurement period. The plots received approximately

65.3 cm of rain during May through August and 137.1 cm for the year compared to year 5 when plots received 20.3 cm and 79 cm for the year. This indicates that when soil moisture is high, treatment effects are less likely to be significant.

Major Leaf Nutrients

Statistics for the effects of treatments on leaf nutrient levels for all tree species are presented in table 2. For northern red oak, there was a significant interaction for N between OM removal and U which showed that N was lower (1.83 percent) for OM_0U_+ than for OM_2U_+ (2.04 percent), but was the same (2.30 percent) for OM_2U_- and OM_0U_- treatments, which had the higher N levels. A similar interaction affected the P concentration. Phosphorus concentrations were highest in U_- plots, but it was slightly lower in OM_0U_+ plots. The interaction between OM removal and soil C on K was the lowering of K concentration in compacted treatments. Calcium was significantly higher in northern red oak leaves for U_- plots than for U_+ plots, but the opposite was true for Mg.

White oak leaf nutrient concentrations (N, P, K, C, and Mg) were significantly affected by U treatments (table 3). The interaction between OM removal and soil C showed a decrease in P and K concentrations for the OM_2C_0 treatment compared to OM_2C_2, but for Ca, treatment combination had the highest concentration. In the three-way interaction, Ca concentrations were higher in the $OM_2C_0U_+$ treatment than for $OM_2C_0U_-$ treatment.

As for shortleaf pine, controlling the U significantly increased N, P, and K concentrations in shortleaf pine needles. There were significant interactions for N, Ca, and Mg concentrations. Leaf N was higher for treatment combinations OM_0C_0 and OM_2C_2 than for OM_0C_2 and OM_2C_0. In the three-way interaction, both Ca and Mg levels were highest for the severe compaction treatment (C_2), but these nutrients either increased or decreased with severe compaction in combination with the different levels of OM removal and U.

Minor and Trace Nutrients

Statistics for leaf concentrations of S, Fe, Mn, Zn, and B, are presented in table 3. For northern red oak, there was an interaction for S between OM removal and U, which showed that S concentrations were highest for the OM_0U_- treatment and progressively decreased for OM_2U_-, OM_2U_+, and OM_0U_+ treatments. The three-way interaction between organic matter removal, compaction, and understory for Zn and B showed that concentrations were highest when the understory was controlled, but was also higher when combined with severe compaction compared to no compaction. The contribution of organic matter removal followed no general pattern.

For white oak, there was an interaction between OM removal and soil C, which showed that the higher levels of Mn were associated with the higher level of compaction (table 3). Interactions for S, Fe, and Zn showed that U and OM removal were major contributors to the concentration level. Mean leaf concentrations of S, Fe, and Zn were lower when OM_2 was part of the treatments compared to OM_0. Also, concentrations were higher when U was absent compared to when the U was present.

For shortleaf pine, there was an interaction for S between OM removal and soil C, which showed the order for treatment combinations was $OM_2C_2 > OM_0C_0 > OM_0C_2 > OM_2C_0$. The effects of U were also significant for Fe. It was higher for U_+ than for U_- treatments (table 3). The significant interaction between soil C and U showed that Mn concentrations were higher for trees with U than without U. The order of treatment combinations for decreasing Mn were in the sequence $C_2U_+ > C_0U_+ > C_0U_- > C_2U_-$. Both Zn and B were affected by a two-way interaction between soil C and U and a three-way between the three treatments. The two-way interaction showed that both Zn and B were higher for treatment combinations C_0U_- and C_2U_+ than for C_2U_- or C_0U_+. The three-way interaction for Zn and B showed that levels were higher for the treatment combination OM_2C_2 than for OM_0C_0 with no clear trend for the contribution of U treatments.

Soil nutrient levels are not presented in this report. However, Johnson and Todd (1998), who investigated the long-term changes in nutrient pools of a mixed oak forest following saw log harvesting vs. whole-tree harvesting, reported that significant differences found in foliar nutrient concentrations were consistent with differences in soil nutrient concentrations. Also, some differences or trends in leaf nutrient concentrations may be due to higher rates of organic matter decomposition from stand residues.

There were few significant leaf nutrient concentrations that could be attributed to main

effects alone (OM removal and soil C). However, soil C can reduce aeration and cause poor root ramification, thus reducing the uptake of nutrients. Limited water and nutrient availability to plants due to compaction are major constraints to plant growth. Youngberg (1959) concluded that soil K and P fixation occurs more readily as moisture availability decreases as reported for this study for soils in C_2 plots.

The big effect that tended to be common for many leaf nutrients was the importance of understory (tables 2 and 3). Leaf nutrient concentrations were not always highest for treatment combinations for U- plots. Weeds reduce the amount of water and nutrients available to trees growing in their vicinity. Understory competition for nutrients and water can increase seedling mortality and decrease foliar nutrient concentrations. Périé and Munson (2000), who evaluated scarification,

Table 2.—*Leaf macro nutrient concentrations and p value if greater than 0.05 for significant interactions for 5-year-old northern red oak, white oak, and shortleaf pine in the Missouri Long-Term Soil Productivity study*

Species	Treatment[1]	Nutrient				
		N	P	K	Ca	Mg
		g/kg				
N. red oak	OM_0	20.80 a[2]	1.09 a	6.31 a	8.88 a	1.90 a
	OM_2	21.51 a	1.18 b	6.18 a	8.20 a	1.95 a
	C_0	21.69 a	1.17 a	6.42 a	8.80 a	1.93 a
	C_2	20.63 a	1.10 a	6.08 a	8.26 a	1.46 a
	U-	22.98 a	1.13 a	6.01 a	9.20 a	1.65 a
	U_+	19.34 b	1.16 a	6.48 b	7.86 b	2.21 b
	OM x C	NS	NS	0.053	NS	NS
	OM x U	0.046	0.022	NS	NS	NS
	C x U	NS	NS	NS	NS	NS
	OM x C x U	NS	NS	NS	NS	NS
White oak	OM_0	19.11 a	1.03 a	5.75 a	9.73 a	1.47 a
	OM_2	18.94 a	1.08 a	5.92 a	10.03 a	1.34 a
	C_0	19.62 a	1.01 a	5.62 a	10.27 a	1.35 a
	C_2	18.43 b	1.10 a	6.05 a	9.50 a	1.56 a
	U-	20.03 a	0.97 a	5.31 a	9.20 a	1.25 a
	U_+	18.01 b	1.14 b	6.35 b	10.57 b	1.56 b
	OM x C	NS	0.049	0.002	0.006	NS
	OM x U	NS	NS	NS	NS	NS
	C x U	NS	NS	NS	NS	NS
	OM x C x U	NS	NS	NS	0.042	NS
Shortleaf pine	OM_0	14.97 a	1.07 a	5.60 a	2.03 a	0.85 a
	OM_2	14.90 a	1.08 a	5.84 a	1.93 a	0.93 a
	C_0	14.52 a	0.94 a	6.43 a	1.76 a	0.72 a
	C_2	14.91 a	1.00 a	6.51 a	2.21 b	1.70 a
	U-	16.57 a	1.12 a	5.99 a	1.90 a	0.86 a
	U_+	13.29 b	1.03 b	5.20 b	2.05 a	0.91 a
	OM x C	0.002	NS	NS	NS	0.023
	OM x U	NS	NS	NS	NS	NS
	C x U	NS	NS	NS	0.004	0.035
	OM x C x U	NS	NS	NS	0.001	0.001

[1]Organic matter removal = bole only (OM_0) and whole tree plus forest floor (OM_2). Soil compaction = none (C_0) and severe (C_2). Understory = understory absent (U-) and understory present (U_+).

[2]Means between treatment levels followed by the same letter are not significantly different from each other ($P \geq 0.05$) within a column and treatment by Tukey's test.

fertilization, and herbicide treatments, applied alone and in combinations on white pine (*Pinus strobus* L.) in Ontario, Canada, reported that many of the effects of understory control that were present at age 4 were present at age 10.

Tree Growth

There were significant interactions between OM removal and soil C for height, height growth, and d.b.h. of northern red oak, and for the height of white oak, but not for shortleaf pine (table 4). Also, there were significant interactions between U and soil C for all species. For northern red oak, height, height growth, and d.b.h. were less in plots that received the combination of OM_0 and C_2 treatments compared to trees in plots receiving OM_0 and C_0 treatments. But height, height growth, and diameter of

Table 3.—*Some minor and trace leaf nutrient concentrations and p value if greater than 0.05 for significant interactions of 5-year-old northern red oak, white oak, and shortleaf pine in the Missouri Long-Term Soil Productivity study*

Species	Treatment[1]	Nutrient				
		S	Fe	Mn	Zn	B
		g/kg		μg/kg		
N. red oak	OM_0	0.124 a[2]	93.17 a	962.54 a	34.95 a	35.75 a
	OM_2	0.117 a	87.17 a	1,200.33 b	35.08 a	32.59 a
	C_0	0.123 a	85.50 a	1,033.17 a	34.26 a	33.54 a
	C_2	0.119 a	94.83 a	1,129.71 a	35.77 a	34.69 a
	U-	0.129 a	109.25 a	1,094.88 a	38.57 a	34.60 a
	U+	0.113 b	71.08 b	1,068.00 a	31.45 b	33.73 b
	OM x C	NS	NS	NS	0.035	NS
	OM x U	0.015	NS	NS	NS	NS
	C x U	NS	NS	NS	NS	NS
	OM x C x U	NS	NS	NS	0.010	0.025
White oak	OM_0	0.116 a	65.63 a	742.46 a	16.70 a	47.17 a
	OM_2	0.123 a	75.38 a	804.96 a	16.63 a	47.37 a
	C_0	0.116 a	67.54 a	679.08 a	16.91 a	43.61 a
	C_2	0.113 a	73.46 a	868.33 b	16.42 a	50.94 a
	U-	0.118 a	68.50 a	618.13 a	18.11 a	50.00 a
	U+	0.121 a	72.50 a	929.29 b	15.23 b	44.54 a
	OM x C	NS	NS	0.029	NS	NS
	OM x U	0.049	NS	NS	NS	NS
	C x U	NS	NS	NS	NS	NS
	OM x C x U	NS	0.051	NS	0.042	NS
Shortleaf pine	OM_0	0.087 a	68.46 a	305.29 a	42.21 a	13.92 a
	OM_2	0.086 a	48.00 a	369.46 a	39.93 a	14.29 a
	C_0	0.088 a	70.79 a	302.08 a	39.84 a	14.97 a
	C_2	0.083 b	45.67 a	372.67 a	42.33 a	3.24 a
	U-	0.092 a	41.92 a	248.29 a	41.33 a	14.75 a
	U+	0.079 b	74.54 b	426.46 b	40.81 a	13.45 a
	OM x C	0.001	NS	NS	NS	NS
	OM x U	NS	NS	NS	NS	NS
	C x U	0.020	NS	0.003	0.001	0.003
	OM x C x U	NS	NS	NS	0.001	0.001

[1]Organic matter removal = bole only (OM_0) and whole tree plus forest floor (OM_2). Soil compaction = none (C_0) and severe (C_2). Understory = understory absent (U-) and understory present (U_+).

[2]Means between treatment levels followed by the same letter are not significantly different (NS) from each other ($P \geq 0.05$) within a column and treatment by Tukey's test.

northern red oak in OM_2 treated plots with either C_0 or C_2 followed the order of OM_0C_0 > $OM_2C_0 \geq OM_2C_2 > OM_0C_2$. The d.b.h. for white oak was largest for trees in the combination OM_0C_2, followed by OM_0C_0, OM_2C_2, and OM_2C_0. The height growth and d.b.h. of northern red oaks in the C_0U_- treatment were largest followed by C_2U_-, C_2U_+, and C_0U_+.

The three-way interaction showed that plots with the treatment combination of no compaction, bole only removal, and no understory

Table 4.—*Planted height and diameter at 1 centimeter above the ground, height and d.b.h. after 5 years, and total growth difference between treatments along with p value if greater than 0.05 for significant treatment interactions for northern red oak, white oak, and shortleaf pine for the Missouri Long-Term Soil Productivity study*

Tree and Treatment [1]	Planted		After 5 years			Treatment difference		
	Ht.	Dia.	Ht.	Ht. growth	d.b.h	Ht.	Ht. growth	d.b.h.
	cm	mm	cm	cm	mm	Percent		
N. red oak:								
OM_0	34.22 a[2]	4.53 a	101.07 a	65.98 a	15.64 a			
OM_2	33.04 a	4.90 a	97.87 a	64.87 a	15.34 a	3	2	2
C_0	34.11 a	4.61 a	103.79 a	70.11 a	15.57 a			
C_2	34.24 a	4.72 a	94.45 b	60.22 b	15.37 a	9	14	1
U_-	33.69 a	4.84 a	104.52 a	70.74 a	18.16 a			
U_+	34.28 a	4.63 a	94.23 b	60.13 b	12.86 b	10	15	29
OM x C	--------	--------	0.0003	0.0435	0.0071			
U x OM	--------	--------	0.0221	NS	NS			
U x C	--------	--------	NS	0.0110	0.0134			
U x OM x C	--------	--------	NS	0.0471	NS			
White oak:								
OM_0	16.56 a	4.53 a	81.76 a	65.10 a	14.75 a			
OM_2	15.93 a	4.44 a	71.34 b	55.13 b	13.46 b	13	15	9
C_0	16.44 a	4.54 a	76.08 a	59.43 a	14.44 a			
C_2	15.96 a	4.50 a	75.87 a	59.70 a	13.63 b	<1	<1	6
U_-	16.27 a	4.48 a	81.42 a	65.02 a	16.81 a			
U_+	16.24 a	4.58 a	70.37 b	53.94 b	11.18 b	14	17	33
OM x C	--------	--------	0.0394	NS	NS			
U x OM	--------	--------	NS	NS	NS			
U x C	--------	--------	NS	NS	0.0021			
U x OM x C	--------	--------	NS	NS	NS			
Shortleaf pine:								
OM_0	15.58 a	3.06 a	188.98 a	170.30 a	38.10 a			
OM_2	14.78 a	3.07 a	185.26 a	165.41 a	35.84 a	2	3	6
C_0	15.06 a	2.75 a	173.02 a	157.32 a	33.07 a			
C_2	14.49 a	2.93 a	200.79 b	185.55 b	40.94 b	14	15	19
U_-	14.87 a	3.01 a	193.87 a	169.61 a	42.84 a			
U_+	15.43 a	3.01 a	180.86 a	173.67 a	31.49 b	7	2	26
OM x C	--------	--------	NS	NS	NS			
U x OM	--------	--------	NS	NS	NS			
U x C	--------	--------	0.0028	0.0019	0.0095			
U x OM x C	--------	--------	NS	NS	NS			

[1]Organic matter removal = bole only (OM_0) and whole tree plus forest floor (OM_2). Soil compaction = none (C_0) and severe (C_2). Understory = understory absent (U_-) and understory present (U_+).

[2]Means between treatment levels followed by the same letter are not significantly different (NS) from each other ($P \geq 0.05$) within a column and treatment by Tukey's test.

($OM_0C_0U_-$) had the most height growth followed by plots with severe compaction, whole tree plus forest floor removal, and no understory ($OM_2C_2U_-$). Plots with the lowest amount of height growth were those with severe compaction and an understory (C_2U_+) plus only boles removed or the whole tree plus forest floor removed.

The effect of the significant interaction for white oak height showed that the tallest trees were in OM_0C_0 plots followed by OM_0C_2, OM_2C_2 and OM_2C_0. White oak trees with the largest d.b.h. were in the C_0U_- and C_2U_- treatment combinations followed by C_2U_+ and C_0U_+. The interaction effects of soil C and U were somewhat different for height and d.b.h. of shortleaf pine. The tallest trees were in treatments receiving the highest level of soil C with and without U control followed by C_0U_- and C_0U_+. But shortleaf pines in C_2U_- treatment had the most height growth, followed by C_0U_-, C_2U_+, and C_0U_+. Treatment combinations with the largest d.b.h. for shortleaf pine were in plots with U control, in the order of C_2U_-, C_0U_-, C_2U_+, and C_0U_+.

Overall, soil C decreased the height and height growth of northern red oak, the d.b.h. of white oak, and increased the height, height growth, and d.b.h. of shortleaf pine. Apparently soil conditions created by soil C were more favorable for the growth of shortleaf pine than for northern red oak or white oak. Perhaps differences in root morphology and physiology of shortleaf pine compared to the oaks could account for some of the better growth.

Controlling the U caused a marked increase in all growth variables for northern red oak and white oak, but only the d.b.h. of shortleaf pine. Undoubtedly, trees benefited from the absence of U. In general, more of the leaf nutrients were higher for all species in U_- plots compared to U_+ plots. However, these nutrient data must be interpreted with some caution because they do not take plant biomass into account, which if considered, could indicate a dilution effect for faster or slower growing trees. For example, white oak grew significantly better in the U_- treatment, but the U_+ treatment had, with few exceptions, higher leaf nutrient concentrations (tables 2 and 3). Foliar nutrient levels were in the sufficiency range for all species (Mills and Jones 1996).

SUMMARY

Differences in soil moisture and temperature, leaf nutrient concentrations, height, height growth, and d.b.h. after 5 years, can be, in part because of significant interactions, attributed to OM removal and soil C treatments. With some exceptions, soil C decreased the height and height growth of northern red oak, the d.b.h. of white oak, and increased the height, height growth, and d.b.h. of shortleaf pine. Apparently soil conditions created by soil C were more favorable for the growth of shortleaf pine than for northern red oak or white oak. Furthermore, these results indicate that although there were indications of soil C and OM removal evident at age 5, the effect of understory vegetation impacted soil moisture and temperature, tree nutrition, and growth as much or more than organic matter removal or soil compaction on this site. The full impact of organic matter removal and soil compaction may not be realized until there is crown closure in these tree stands.

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RELATIONSHIPS BETWEEN BIOTIC AND ABIOTIC FACTORS AND REGENERATION OF CHESTNUT OAK, WHITE OAK, AND NORTHERN RED OAK

Songlin Fei, Kim C. Steiner, James C. Finley, and Marc E. McDill[1]

ABSTRACT.—A series of substantial field surveys of 38 mixed-oak stands in central Pennsylvania were carried out during 1996–2000. All the stands were surveyed 1 year prior to harvest, and 16 stands have been surveyed 1 year after harvest. Three abiotic factors at stand scale, four abiotic factors at plot scale, and two biotic factors and one abiotic factor at subplot scale was used to analyze their relationships to the abundance and composition of regeneration. Stand scale factors were the most influential factors on oak regeneration. Stand aspect together with slope position explained most of the variation in abundance of both advanced and post-harvest regeneration. Plot slope shape and exposure angle played an important position in oak regeneration. All biotic factors at subplot scale had a strong correlation with regeneration abundance. In general, light factors (stand aspect and exposure angle), other physical conditions (slope position and slope shape), and canopy composition immediately above the plot played strong roles in regeneration in these mixed-oak stands.

Throughout eastern Northern America, natural regeneration of oaks is often difficult to obtain even where oaks are dominant components in the overstory before harvest. Although the problem is widespread both geographically and by species, there is no apparent universal solution. The major causes of regeneration difficulty may change from site to site and region to region. Because of this variability, we need to view "the oak regeneration problem" as having both local and regional aspects (Lorimer 1992). As Crow (1988) has pointed out for northern red oak, specific prescriptions for local conditions are needed to supplement general guidelines for regenerating oak.

In this paper, we describe some results pertaining to the relationship between oak regeneration and stand conditions from field surveys of 38 mixed-oak stands in Pennsylvania carried out during 1996 to 2000. All the stands were surveyed 1 year prior to harvest, and 16 stands have also been surveyed 1 year after harvest. In each stand, depending on the stand size, 15 to 30 permanent plots with a radius of 26.3 feet (20th-arce plots) were spaced systematically in a square grid to represent the whole stand. Four permanent subplots with a radius 3.72' (milacre plots), were set up within each plot at 16.5 feet from the 20th-acre plot center along each cardinal direction. All together, we measured 4,012 subplots before harvest and 1,600 subplots 1 year after harvest.

Regeneration density and size were sampled in each subplot by species. Biotic and abiotic factors were also recorded at the stand, plot, and subplot level. The following biotic and abiotic factors were measured in the field survey:

- Stand scale: elevation, slope position, and slope aspect;
- Plot scale: slope shape (sum of percentage slope uphill, down hill, and at 90° to aspect; + values = concavity; - values = convexity), slope percentage, exposure angle (the angle between the visible east horizon and west horizon), and slope aspect;
- Subplot scale: micro-topography (mound, slope, pit, and flat), closest canopy species, distance to closest canopy tree, largest canopy species over plot, and distance to largest canopy tree.

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Oak regeneration quantity and quality (size) were both very important to ensure successful oak regeneration, but in this paper we mainly focused on regeneration. Analysis of variance was performed in SAS by using a linear nested model to examine relationships between the above variables and regeneration density of chestnut oak (*Quercus prinus* L.), northern red oak (*Q. rubra* L.), and white oak (*Q. alba* L.).

Before harvest, stand basal area for all the stands surveyed ranged between 72 and 130 ft^2/ac, with oak species representing 31 to 91 percent of the total (typically >50 percent). After modified either by a clearcut or shelterwood harvest, the basal area of the remaining trees ranged from 7 to 52 ft^2/ac, and most of the remaining overstory trees were oaks. Over all of the surveyed stands, the major oak regeneration species for both pre-harvest and post-harvest assessment were chestnut oak (3,004 and 4,467 stems/acre before and after harvest, respectively), northern red oak (2,149 and 1,237 stems/acre, respectively) and white oak (799 and 489 stems/acre, respectively).

Red maple (*Acer rubrum* L.) was the most abundant regeneration species (21,462 and 14,613 stems/acre before and after harvest). Sweet birch (*Betula lenta* L.), although not widely distributed all over the region, had occurrences of regeneration density as high as 8,300 stems/acre in several stands after harvest. Since not all the stands have yet been surveyed 1 year after harvest, the sample sizes are different before and after harvest. Hence it is not quite appropriate to compare regeneration abundance before and after harvest because of possible sample bias. Nevertheless, as a trend, average regeneration density decreased after harvest for red maple, northern red oak, and white oak, but increased for chestnut oak.

The results of linear nested models showed that stand level attributes were the most influential factors affecting oak regeneration abundance. In all the nested models, site always had a significant influence on the regeneration density of all three oak species. ANOVA of oak regeneration density by the stand scale factors showed that stand slope position combined with stand aspect had a significant influence on regeneration density of all three oak species. Aspect and slope position were two most important abotic factors in oak regeneration because they not only represented stand's physiography, but also reflected soil, moisture, and light conditions of the stand (Finney and others 1962, Kercher and Goldstein 1977, Standiford and others 1997).

Similar to former research results (McCarthy and others 1984, Seischab 1985), the distribution of regeneration by stand aspect showed that stands with northern aspects had the lowest regeneration density for all three oaks in both time periods except white oak after harvest (table 1). Regeneration densities of chestnut oak on the northern aspects were significantly lower than the other aspects.

The distribution of regeneration by slope position (table 2) showed that white oak advance regeneration appeared to be favored by flat slope position (i.e., ridges and bottoms), while chestnut oak appeared to be favored by upper slope locations. Northern red oak regeneration did not differ significantly among slope positions. Red maple had the most abundant regeneration density across the whole region, and it regenerated best on northern aspects and on benches.

At the 20th-acre plot level, exposure angle and slope shape were two of the most important factors affecting oak regeneration. In all cases, small exposure angle limited oak regeneration

Table 1.—*Regeneration density (stems/acre) versus stand aspect*

Stand aspect	Red maple		White oak		Chestnut oak		Northern red oak		Total oak	
	Pre-	Post-	Pre-	Post-	Pre-	Post-	Pre-	Post-	Pre-	Post-
N (6,5)*	24999 a**	10137 a	790 a	1660 a	1533 a	2071 a	961 a	412 a	3284 a	4143 a
E (7,2)	22383 a	10165 a	945 a	2490 a	3753 b	5198 b	3189 a	2245 a	7887 b	9933 b
S (15,8)	21129 a	16171 a	1602 a	615 a	3814 b	5895 b	1893 a	1162 a	7309 b	7672 b
W (8,3)	18458 a	15196 a	3384 a	183 a	5889 b	4743 b	1910 a	1592 a	11183 c	6518 b

* Number of stands that have this stand aspect before and after harvest.
** Mean densities within the same column not sharing same letters are significantly different at $P < 0.05$.

Table 2.—*Regeneration density (stems/acre) versus slope position*

Slope position	Red maple		White oak		Chestnut oak		Northern red oak		Total oak	
	Pre-	Post-	Pre-	Post-	Pre-	Post-	Pre-	Post-	Pre-	Post-
Ridge (6, 2)*	24601 a**	24144 a	6459 a	162 a	4423 a	2176 a	2882 a	1233 a	13764 a	3571 a
Upper slope (6, 3)	20510 a	10379 a	325 b	—	7096 b	9693 b	1320 a	1836 a	8741 ab	11529 b
Mid slope (9, 5)	22480 a	13182 a	702 b	1576 a	2570 a	5712 a	1573 a	689 a	4845 b	7977 a
Lower slope (8, 8)	12280 a	12632 a	539 b	982 a	3098 a	2779 a	1937 a	1151 a	5574 b	4912 a
Bench (2, 0)	57600 b	—	611 b	—	3208 a	—	1405 a	—	5224 b	—
Plateau (5, 0)	17057 a	—	—	—	—	—	4691 a	—	4691 b	—
Bottom (2, 0)	19800 a	—	4584 a	—	4508 a	—	1421 a	—	10513 ab	—

* Number of stands that have this slope position before and after harvest.
* Mean densities within the same column not sharing same letters are significantly different at $P < 0.05$.

density (figs. 1 to 3). Alternatively, presumably because of other limiting factors, large exposure angle did not always mean high density of oak regeneration. But if other factors were not limiting, both white oak and northern red oak had optimum exposure angles over 140 degrees for advanced regeneration (figs. 1 and 3), while chestnut oak had optimum exposure angles over 110 degrees (fig. 2). It appears that the minimum exposure angle for abundant regeneration may be lower for chestnut oak than for white oak and northern red oak.

As illustrated by figures 4 through 6, all oaks had the most abundant regeneration with near linear slope shape, and had the lowest density either with extreme concave or convex shapes. Regeneration abundance declined abruptly as slopes became noticeably concave or convex. Exposure angle and slope shape had the similar affects on post-harvest regeneration abundance.

Neither micro-topography nor distance to the closest tree and largest canopy tree showed significant influences on oak regeneration, but there were strong relationships between regeneration and the species of the closest or largest canopy tree. Not surprisingly, the three oak species had the most abundant regeneration density when the closest or largest canopy trees were of their own species (table 3). White oak was only marginally more abundant when the closest tree was white oak as opposed to black gum (*Nyssa sylvatica* Marsh), but otherwise regeneration density was approximately doubled or tripled when the closest tree species is of its own kind. Pignut hickory (*Carya glabra* Sweet), also, tended to be associated with high oak regeneration. On the other hand, sweet birch as a closest tree species in the canopy was constantly associated with poor oak regeneration. The relationships between regeneration density and largest canopy tree were similar.

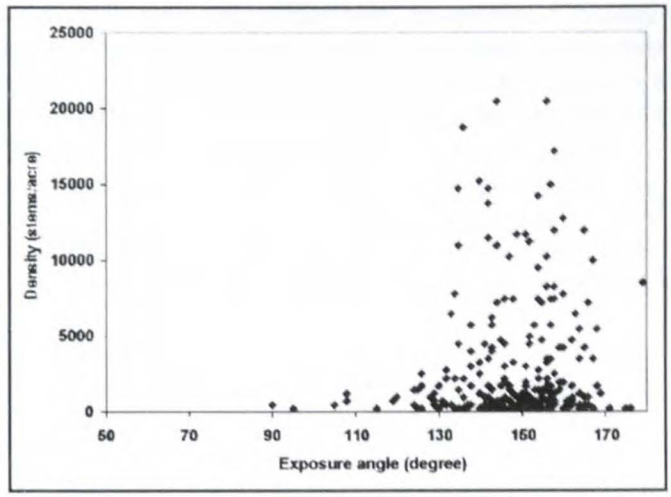

Figure 1.—*Density of advanced regeneration of white oak (stems/acre) versus plot exposure angle at plot level (only plots with at least one white oak seedling were included, n=272).*

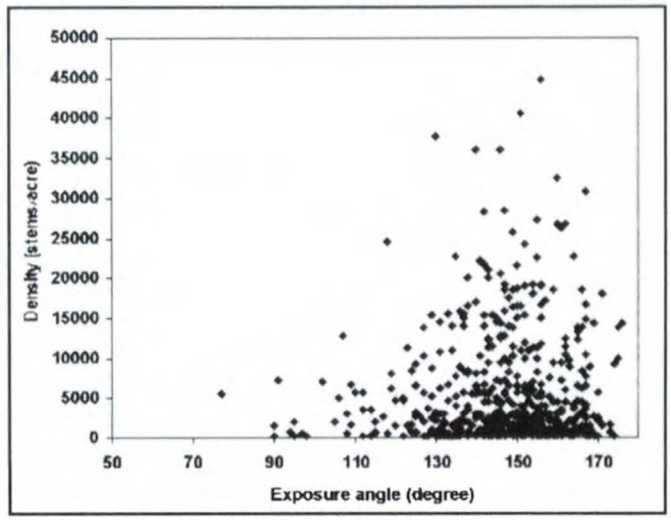

Figure 2.—*Density of advanced regeneration of chestnut oak (stems/acre) versus plot exposure angle at plot level (only plots with at least one chestnut oak seedling were included, n=558).*

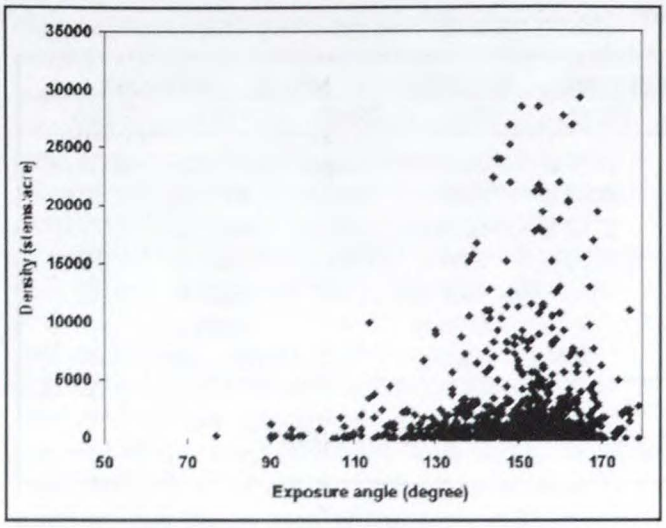

Figure 3.—*Density of advanced regeneration of northern red oak (stems/acre) versus plot exposure angle at plot level (only plots with at least one northern red oak seedling were included, n=710).*

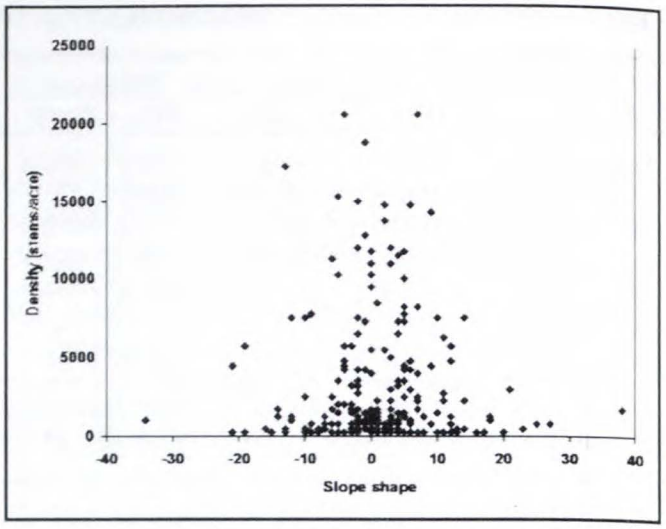

Figure 4.—*Density of advanced regeneration of white oak (stems/acre) versus slope shape at plot level (only plots with at least one white oak seedling were included, n=272).*

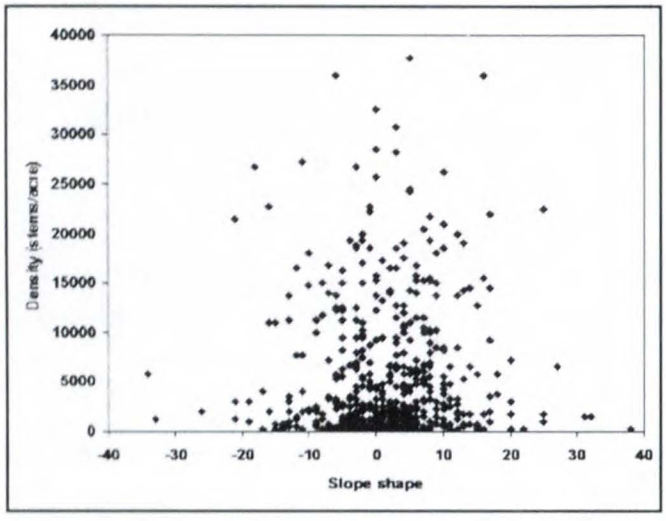

Figure 5.—Density of advanced regeneration of chestnut oak (stems/acre) versus slope shape at plot level (only plots with at least one chestnut oak seedling were included, n=558).

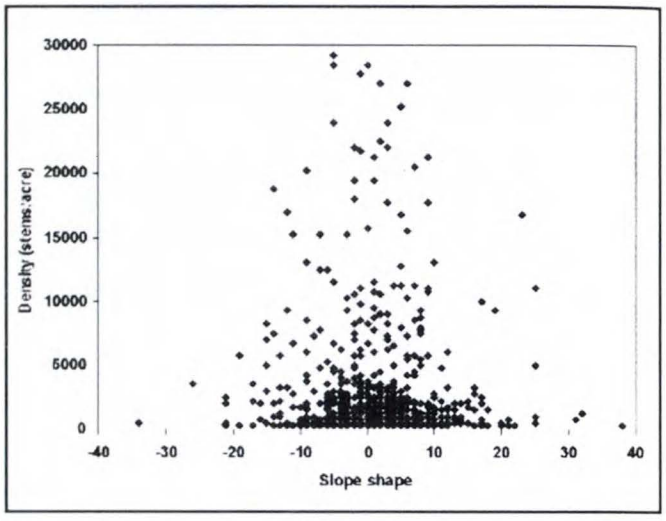

Figure 6.—*Density of advanced regeneration of northern red oak (stems/acre) versus slope shape at plot level (only plots with at least one northern red oak seedling were included, n=710).*

Table 3.— *Advanced oak regeneration density versus closest tree species to subplot.*

White oak			Chestnut oak			Northern red oak		
Closest tree species	Density (stems/acre)		Closest tree species	Density (stems/acre)		Closest tree species	Density (stems/acre)	
White oak	5,692	a*	Chestnut oak	9,096	a	Northern red oak	6,349	a
Black gum	4,824	a	Black oak	4,010	b	Pignut hickory	1,944	b
Red maple	2,690	b	Pignut hickory	3,783	b	Black oak	1,811	b
Pignut hickory	2,125	b	Black gum	3,591	b	Scarlet oak	1,667	b
White pine	2,125	b	Tulip tree	3,571	b	Red maple	1,620	b
Black oak	1,818	b	Northern red oak	3,339	b	White oak	1,413	b
Scarlet oak	1,202	bc	Scarlet oak	3,066	b	White pine	1,148	bc
Chestnut oak	1,151	bc	Red maple	2,994	b	Chestnut oak	1,118	bc
Northern red oak	911	bc	White oak	2,595	b	Black gum	618	c
Sweet birch	429	c	White pine	2,438	b	Tulip tree	577	c
Tulip tree	143	c	Sweet birch	694	c	Sweet birch	528	c

* Mean densities within the same column not sharing same letters are significantly different at $P < 0.05$.

The following are our conclusions:

- In Pennsylvania, slope position together with the stand aspect were the most important factors influencing oak regeneration.
- Exposure angle (open sky between visible eastern and western horizon) was influential on oak regeneration. Large exposure angles did not appear to inhibit oak regeneration. However, small angles did, and this lower limit appeared to vary among species. Different oaks had different optimum exposure angles. The optimum exposure angles for white oak and northern red oak were over 140 degrees, while for chestnut oak the optimum was over 110 degrees.
- Linear or only slightly convex or concave slope shapes appeared to favor oak regeneration.
- There were significant relationships between the species of closest or largest canopy tree over subplots and the abundance of oak regeneration. Not surprisingly, oak regeneration was strongly favored by the presence of the same species in the canopy above the plot. The presence of pignut hickory as the closest tree species also favored oak regeneration, while regeneration was the lowest when the closest tree was sweet birch.

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SITE AND CANOPY CHARACTERISTICS ASSOCIATED WITH OAK ADVANCE REPRODUCTION IN MATURE OAK-HICKORY FORESTS IN THE RIDGE AND VALLEY PROVINCE IN TENNESSEE

Leslie S. Chadwell and David S. Buckley[1]

ABSTRACT.—To investigate hypotheses regarding effects of competitors and site quality on oak regeneration, we documented site factors and oak seedling composition, size, and abundance in the Ridge and Valley Province of Tennessee. Small oak seedlings were most abundant on productive soils and mesic landform positions, whereas large oak seedlings were most abundant on less productive soils and drier landform positions. Patterns in the abundance of large oak seedlings and saplings were consistent with the hypothesis that oaks are most competitive on drier and poorer sites. Results were mixed with regard to existing hypotheses concerning particular competitors and competition for light.

Reduced oak regeneration in the eastern U.S. remains a significant problem despite more than two decades of research on potential causal factors and solutions. Most oak regeneration research has focused on the evaluation of different management techniques for releasing existing natural oak advanced reproduction, or on increasing the success and competitive potential of planted oaks (Beck 1970, Sander 1972, Loftis 1990, Buckley and others 1998). Fewer studies have investigated the underlying factors that affect the presence, absence, and abundance of oak advanced reproduction available for subsequent management on sites with little disturbance (Weitzman and Trimble 1957, Carvell and Tryon 1961, Lorimer and others 1994).

Oak forest types and mixtures of oaks and other species are considered the most abundant forest types in the U.S., but forest management techniques used over the past 50 years have favored species that are more shade tolerant or faster growing than oak (Clark 1993). Tennessee forests are 71 percent oak-hickory forest type, 12 percent oak-pine forest type, and 4 percent oak-gum-cypress forest type (Schweitzer 2000). Since forest types containing oak cover 87 percent of the forested land in Tennessee, a continued lack of oak regeneration will cause changes to forest composition that will negatively impact the forest products industry and wildlife. Little information has been published on the severity of the oak regeneration problem in Tennessee. To our knowledge, this is the first study of this kind to take place in the Ridge and Valley Province of Tennessee.

Results of previous research suggest that drought, fire suppression, competition, increased consumption of acorns by wildlife, defoliation, and various soil properties have contributed to oak regeneration failures (Lorimer 1989, LeBlanc 1998, Cook and others 1998, Rogers and Johnson 1998). McGee (1984) found that on average and productive sites, established oak seedlings in the understory were unable to capture small or large canopy gaps. Due to increased competition with species such as red maple (*Acer rubrum* L.), sugar maple (*A. saccharum* Marsh.), yellow-poplar (*Liriodendron tulipifera* L.), and beech (*Fagus grandifolia* Ehrh.), it is uncertain whether oaks will remain the dominant species in hardwood forests in the future.

In the northern Appalachian hardwoods of West Virginia, oak regeneration tends to be more widespread on intermediate to poor sites, with a

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decreasing proportion of oak with increasing site quality (Weitzman and Trimble 1957). Thus, oak species may be more competitive on intermediate and poor sites where competitors are limited (Lorimer 1993). Soil texture, litter depth, pH, and soil moisture may not directly lead to regeneration declines, but they do have an effect on the germination, growth, and mortality of hardwood species (Meiners and others 1984, Meredieu and others 1996).

Past research has generated two primary hypotheses concerning oak regeneration. The first is that the amount of oak reproduction should increase with reduced abundance of potential competitors. The second hypothesis is that oak regeneration should be more successful on drier, poorer soils, warmer aspects, and upper slope positions where competitors are limited by growing conditions. The overall objectives of this study were:

1) to quantify natural oak regeneration in the Ridge and Valley Province of Tennessee in untreated, closed-canopy forests, and
2) to examine effects of competition and site factors associated with greater abundance of oak reproduction.

METHODS

Stand Types and Study Region

We sampled 33 mature mixed oak/hardwood stands within five counties of the Ridge and Valley Physiographic Province in Tennessee, also known as The Great Valley of East Tennessee. All sampled stands had oak as the dominant canopy species and had not been significantly disturbed (canopies have remained intact) within the last 50 years. Highgrading is common in the region, however, and the possibility of past highgrading of some of the stands sampled cannot be ruled out. The Ridge and Valley Province constitutes 18 percent of the total land area in Tennessee (Springer and Elder 1980). The region is characterized by its folded rock formations oriented in a northeast-southwest direction, comprising nearly 5 million acres of land.

There are seven soil provinces within the region, each further divided into soil associations (table 1). Three soil provinces were not sampled because of limited forest cover. There were a total of eight different soil associations within the four soil provinces sampled. Two of the eight soil associations were not sampled due to soils such as calcareous shales and calcareous sandstones that were not characteristic of the region. Soils within the soil associations included in the study ranged from moderately high productivity to strongly acidic soils low in fertility (Springer and Elder 1980).

Transect Design and Measurements

For sampling, belt transects 60 m long and 5 m wide were placed parallel to the contour of the stand (fig. 1). Three landform positions (southeast mid-slope, northwest mid-slope, and ridgetop) were sampled within each soil association. Each of these three landform positions was

Table 1.—*Comparison of the soil associations sampled in the Tennessee portion of the Ridge and Valley Physiographic Province. Adapted from Springer and Elder (1980) and Owenby and Ezell (1992).*

Soil Association	Parent Material	Landscape	Soil Productivity	AnnualP recip. (in)	Temp.* (°C)
Fullerton-Dewey	Dolomitic Limestone	Hilly and rolling	Mod. High	53	1 I 24
Fullerton-Bodine	Dolomitic Limestone	Hilly	Strongly acid; low fertility	54	2 I 24
Dunmore-Dewey	Dolomitic Limestone	Rolling and hilly	Moderate	42	1 I 24
Talbott-Rock-Etowah	Clayey Limestone	Undulating to hilly	Moderate	55	3 I 23
Litz-Sequoia-Talbott	Shale	Undulating and rolling	Mod.; lots of coarse fragments	55	3 I 23
Wallen-Talbott-Montevallo	Sandstone, Shale, & Dolomitic Limestone	Steep ridges and rolling valleys	Low; shallow soil	53	3 I 24

* First number is mean temperature in January, second number is mean July temperature.

sampled in two geographic locations within the same soil association for replication. The southeast or northwest-facing aspects coincide with the dominant northeast-southwest orientation of the ridges in the Ridge and Valley Province.

Oak reproduction was sampled within three 5 m x 20 m subplots along the length of each transect. Species and size class were tallied for all oak seedlings and saplings within each subplot. The four size classes used were defined as

1) 0 to 25 cm tall;
2) 26 to 50 cm tall;
3) 51 to 150 cm tall; and
4) greater than 150 cm tall, but less than 8 cm dbh.

Variables measured within transects included slope; percent cover of understory herbs, shrubs, and tree seedlings; litter depth; litter composition; basal area; density of non-oak tree saplings greater than 150 cm tall, but less than 8 cm dbh; percent full photosynthetically active radiation (PAR); the number, composition, and width of canopy layers; and percent canopy cover. All site factor measurements, except for percent full PAR, density of non-oak saplings, and canopy cover percent, were recorded at three sampling points located at 10 m, 30 m, and 50 m down the center of the transect (fig. 1). Percent full PAR and canopy cover percent were recorded every 10 m along the center of the transect. Non-oak sapling density was measured within the three 5 m x 20 m subplot areas along the transect.

We estimated cover of understory herbs, shrubs, and tree seedlings less than 1 m tall within 1 m^2 quadrat frames located on transect sampling points. The species comprising canopy strata over each sampling point were recorded and the thickness of each stratum was quantified using a height pole and clinometer to measure the height of the top and bottom of each stratum > 1 m thick. For analysis, the canopy profile was separated into two sections, upper canopy (> 13 m above the ground) and mid-canopy (4 to 13 m above the ground). Litter depth was measured to the nearest centimeter at each sampling point. Basal area of the stand was determined at each sample point with a 10 BAF prism. The species and diameter at breast height for all 'in' trees were also recorded. Importance values (relative density + relative frequency + relative dominance) for each species were calculated from basal area data for each transect as described in Krebs (1985).

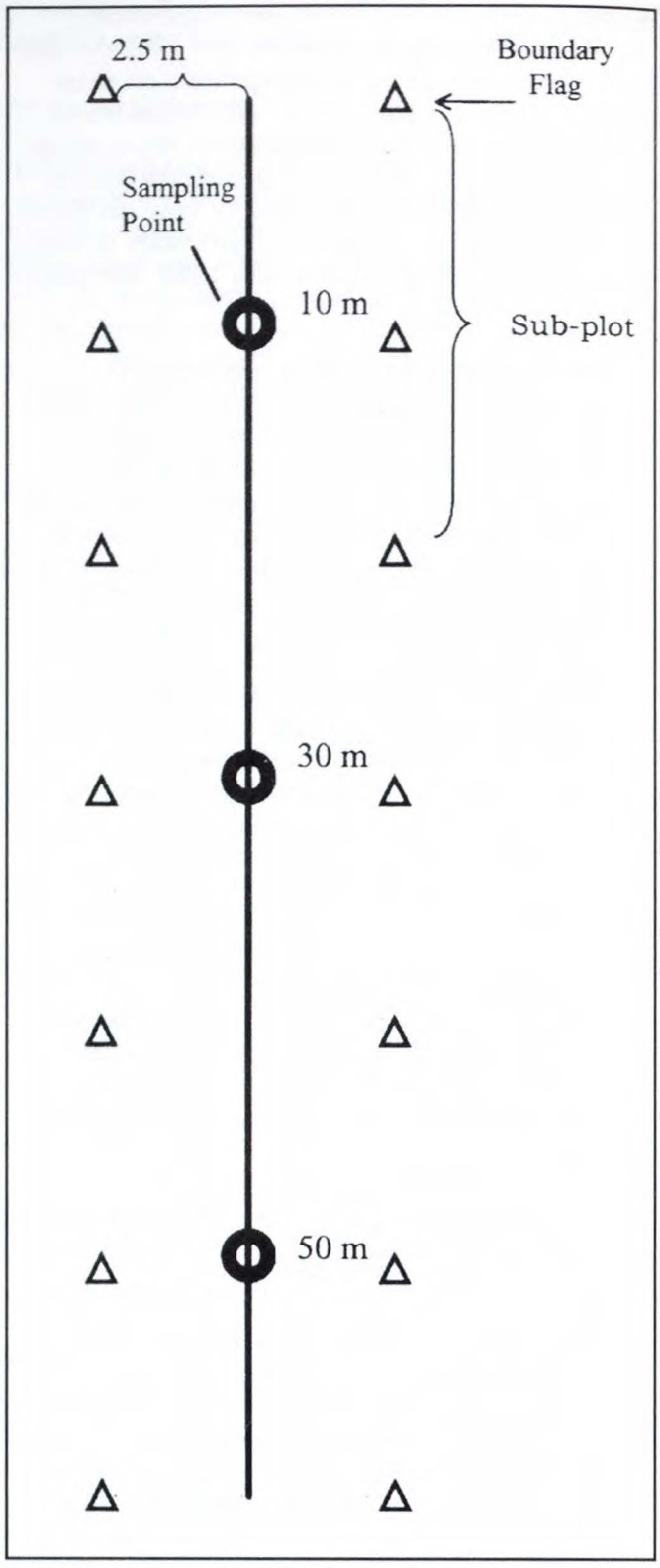

Figure 1.—*Transect design placed parallel to the contour at each sampling site, used to sample stand and oak measurements on the different soil associations and landform positions.*

PAR consists of all wavelengths of light that are used in photosynthesis, and was measured with an AccuPAR Ceptometer (Decagon; Pullman, WA). PAR measurements were taken 1 m above each sampling point along a given transect between 11:00 AM and 3:00 PM EDST. All PAR measurements were taken on clear days, and reference measurements were taken in completely open areas before and after sampling transects. The ratio of below canopy PAR to mean PAR in the open was used to calculate percent full PAR. Percent canopy cover was measured with a CI-110 Digital Plant Canopy Imager (CID, Inc.; Vancouver, WA), which measured the fraction of sky that was visible from beneath the canopy. The Plant Canopy Imager was held 1 m above each sampling point while facing north.

Preliminary analysis indicated that landform positions and soil associations were important across all species and size classes. Thus, analyses were run within landform position-soil association combinations. Regeneration data were averaged over each transect and analyzed with R^2 Variable Selection, Response Surface Analysis, and General Linear Model (GLM) analysis (CD version 8, SAS Institute, Cary, NC). Transects were assumed to be independent ecological units. GLM was used as a final step in the analysis because landform position and soil association were class variables and were not appropriate for variable selection or Response Surface analysis. Separate models were developed for each oak species due to differences in their habitat requirements, as well as for each size class of oak due to potential differences in requirements and interactions at different developmental stages.

RESULTS

Eight species of oak were encountered: white oak (*Quercus alba* L.), chestnut oak (*Q. Montana* Willd.), black oak (*Q. velutina* Lam.), northern red oak (*Q. rubra* L.), scarlet oak (*Q. coccinea* Muenchh.), southern red oak (*Q. falcate* Michx.), post oak (*Q. stellata* Wangenh.), and chinkapin oak (*Q. muehlenbergii* Engelm.). Post oak and chinkapin oak were omitted from individual species analyses because of low abundance and spotty distribution, but were included in analysis of all oak species and size classes combined. Scarlet and southern red oaks have similar habitat requirements and were less abundant. Thus, they were analyzed as a combined group.

The white oak group represented 76 percent of the total number of oak seedlings tallied in the region (table 2). The smallest size class of seedlings comprised 88 percent of the oak seedlings tallied, while the largest size class comprised only 0.5 percent. By far, white oak and chestnut oak were the most abundant species in the 0 to 25 cm and 26 to 50 cm size classes. White oak, chestnut oak, black oak, and northern red oak in the 51 to 150 cm and > 151 cm, but less than 8 cm dbh size classes were fairly comparable. Only two individuals in the scarlet/southern red oak group were found in the 51 to 150 cm size class, and no scarlet/southern red oak individuals were tallied in the > 151 cm, but less than 8 cm dbh size class.

The 12,220 oak seedlings tallied in the 0.99 ha total area sampled suggests densities of 12,341 seedlings/ha, which exceeds the 9,100 seedlings/ha average reported previously for the Southern Appalachians (Cook and others 1998). Only 61, 0.5 percent, of the total numbers of seedlings sampled, however, were greater than 150 cm in height. This suggests a density of 62 seedlings/ha taller than 150 cm, which is well below the previously reported, tentative estimate of 1,000+ seedlings/ha that exceed 140 cm in the Southern Appalachians (Cook and others 1998).

Table 2.—*Total number of oak seedlings tallied by species and size class.*

Species	Size class*				Percent
	1st	2nd	3rd	4th	
White oak	1675	318	49	11	16.8
Chestnut oak	6742	425	69	10	59.3
Black oak	1034	197	49	23	10.7
N. red oak	806	220	65	17	9.1
Scarlet/S. red oak	458	50	2	0	4.2

*1st = seedlings 0 – 25 cm tall, 2nd = seedlings 26-50 cm tall, 3rd = seedlings 51-150 cm tall, and 4th = seedlings 151 cm tall up to 8 cm dbh.

Soil association was a significant variable ($P < 0.05$) in at least one size class for all oak species. Consequently, soil association was included as a variable in all subsequent models developed. Similarly, landform position was found to be significant for most of the oak species and was also included in all models. Small seedlings (up to 50 cm tall) were most abundant on the Fullerton-Dewey, Wallen-Talbott-Montevallo, and Talbott-Rock-Etowah soil associations (fig. 2), which spanned the range of soil productivity (table 1).

In contrast, large seedlings (from 51 cm tall up to 8 cm dbh) were most abundant on the Litz-Sequoia-Talbott and Dunmore-Dewey soil associations (fig. 2), which have soils generally ranked as moderate in productivity (table 1). Mean percent full PAR and mean overstory basal areas varied little across soil associations, and showed no clear relationships to the number of small or large oak seedlings (fig. 2).

Small seedlings were most abundant on the northwest aspect/mid-slope position, whereas large seedlings were most abundant on the southeast aspect/mid-slope position (fig. 3). However, differences in the mean number of large seedlings between landform positions were small due to the low numbers of large seedlings. Changes in mean percent full PAR and mean basal area were slight across landform positions, but mean PAR was greatest on the southeast aspect/mid-slope position and least on the northeast aspect/mid-slope position. Mean basal area was lowest on the southeast aspect/mid-slope position. Thus, small seedlings were most abundant on the landform position with high basal area and the lowest mean PAR, whereas large seedlings were most abundant on the landform position with the greatest mean PAR and lowest mean basal area.

Three-dimensional plots of the abundance of individual size classes within species (not presented) indicated a concentration of seedlings 51 to 150 cm tall and 151 cm tall up to 8 cm dbh for all species but scarlet/southern red oak on the southeast aspect/mid-slope and ridge top landform positions and the Litz-Sequoia-Talbott soil association. Models developed for all species and size classes of oak seedlings combined indicated positive relationships between the number of oak seedlings and the importance value of chestnut oak and the number of oak seedlings and the mean depth (combined thickness of all strata > 1 m thick) of the mid-canopy (table 3). A negative relationship was

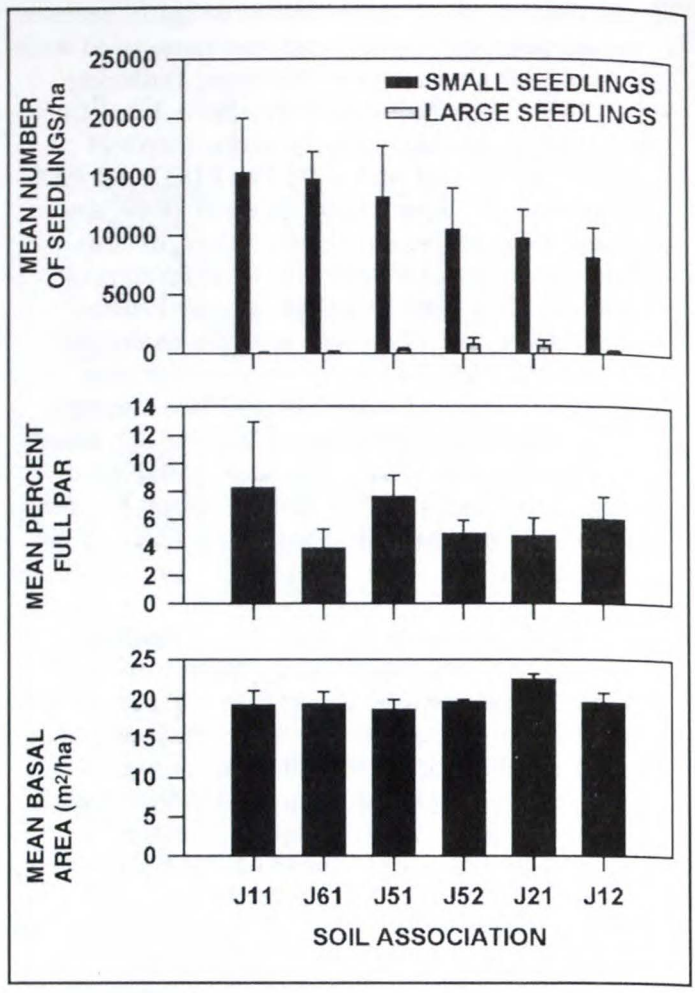

Figure 2.—*Mean number of oak seedlings, mean percent full PAR, and mean overstory basal area by soil association. Small seedlings represent oak seedlings of all species measuring 1 - 50 cm in height and large seedlings represent seedlings of all species measuring 51 cm tall up to 8 cm dbh. For soil associations, J11 = Fullerton-Dewey, J61 = Wallen-Talbott-Montevallo, J51 = Talbott-Rock-Etowah, J52 = Litz-Sequoia-Talbott, J21 = Dunmore-Dewey, J12 = Fullerton-Bodine. Error bars represent 1 standard error.*

indicated between the number of oak seedlings and the number of hickory seedlings 151 cm tall up to 8 cm dbh.

Models for each individual oak species (all size classes combined) indicated positive relationships between the number of seedlings and the importance value of mature oaks of the same species (table 3). The number of white oak seedlings also showed a positive relationship with the importance value of beech and the depth of the canopy. The depth of the canopy was positively related to the number of chestnut oak seedlings, and other positive relationships

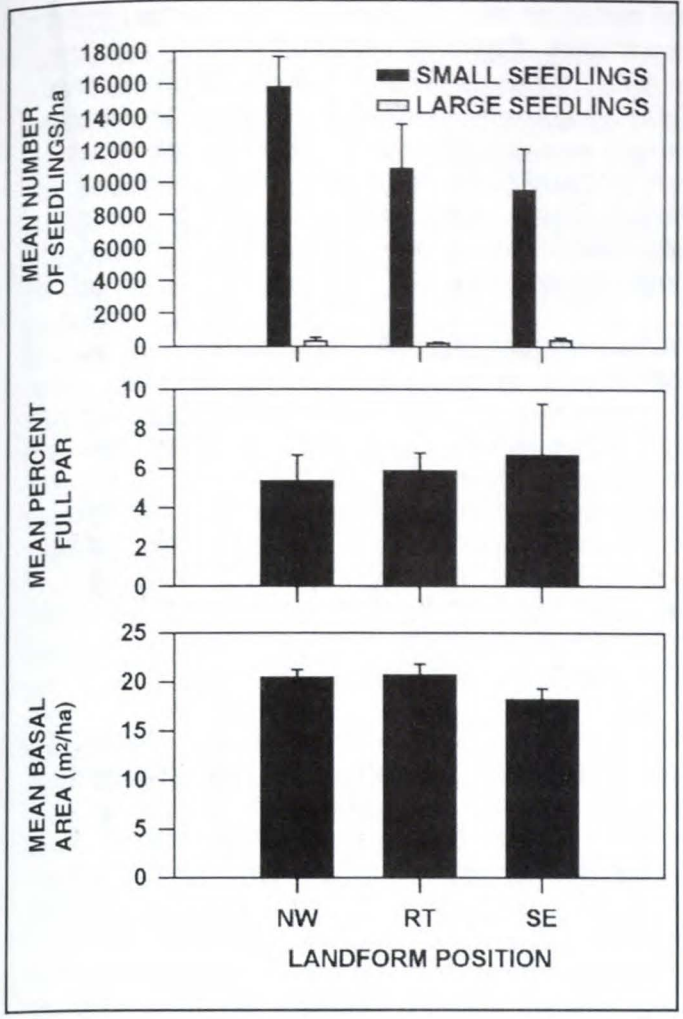

Figure 3.—*Mean number of oak seedlings, mean percent full PAR, and mean overstory basal area by landform position. Small and large seedlings and error bars as in figure 2. NW = northwest aspect, mid-slope position, RT = ridge top, and SE = southeast aspect, mid-slope position.*

were indicated between the numbers of chestnut oak, black oak, and northern red oak seedlings and potential competitors such as blackgum (*Nyssa sylvatica* Marsh.) seedlings and mature pines (table 3). Negative relationships were indicated between percent full PAR and the number of chestnut oak and scarlet/southern red oak seedlings.

Negative relationships between potential competitors and the number of oak seedlings were indicated for chestnut oak, northern red oak, and scarlet/southern red oak (table 3). These potential competitors included sassafras (*Sassafras albidum* (Nutt.) Ness), mature white oak, and mature black oak.

Additional models developed for individual size classes within oak species (not presented) indicated negative relationships between dogwood (*Cornus florida* L.) and the numbers of chestnut oak, black oak, and northern red oak, and between mature maples (red and sugar maple combined) and scarlet/southern red oak. A negative relationship was also indicated between the number of 151 cm tall up to 8 cm dbh northern red oak seedlings and the amount of canopy cover. Finally, a positive relationship was indicated between percent slope and the number of black oak seedlings 151 cm tall up to 8 cm dbh.

R^2 variable selection indicated percent cover of understory herbs, shrubs, and tree seedlings less than 1 meter, litter depth, and litter were not significantly related to the amount of oak regeneration that occurred on the site ($P > 0.05$).

Table 3.— *Models and variables explaining the number of oak seedlings of all species and size classes combined and the number of seedlings of each species (all size classes combined). Signs indicate positive or negative relationships between variables and numbers of oak seedlings. Y-intercepts in the models vary across landform position/soil association combinations. Variable coefficients and Y-intercepts have been omitted for the purpose of efficient presentation.*

Species	Model	R^2 (Percent)	P-value
All Species	LFP[1] Soil[2] +Coiv[3] +Midcan[4] -Hick[5]	54.2	0.0297
White oak	LFP Soil +Woiv[6] +Bchiv[7] +Candep[8]	86.9	<0.0001
Chestnut oak	LFP Soil +Coiv -PAR[9] +Candep +Blgm[10] -Sass[11]	69.6	0.0041
Black oak	LFP Soil +Boiv[12] +Pineiv[13]	73.2	<0.0001
N. red oak	LFP Soil +Nroiv[14] -Woiv +Blgmiv[15] -Sassiv[16] -Midcan	61.0	0.0280
Scarlet/S. red oak	LFP Soil +Scsroiv[17] -Boiv -PAR	71.1	0.0005

[1]Landform position; [2]Soil association; [3]Chestnut oak importance value; [4]Average midcanopy depth (m); [5]Hickory saplings, 4th size class; [6]White oak importance value; [7]Beech importance value; [8]Average total canopy depth (m); [9]Average percent full PAR; [10]Blackgum saplings, 4th size class; [11]Sassafras saplings, 4th size class; [12]Black oak importance value; [13]Pine importance value; [14]Northern red oak importance value; [15]Blackgum importance value; [16]Sassafras importance value; [17]Scarlet and Southern red oak importance value.p

DISCUSSION

The dominance of small seedlings in the white oak group on the study sites may have been due to greater seedling shade tolerance in white oak and chestnut oak than in the red oaks. In hardwoods, the shade tolerance of younger seedlings is often greater than that of older saplings (Burns and Honkala 1990). An alternative explanation is that red oak seed sources may have been reduced by past highgrading, which tended to remove the more commercially important species in the red oak group. A final possibility is that conditions and seed crops in recent years were more favorable for establishment of species in the white oak group than in the red oak group. Provided that size can be reasonably correlated with age, the comparable number of all species except scarlet/southern red oak in the large size classes suggest that long-term survival of white, chestnut, black, and northern red oak measuring 151 cm tall and up to 8 cm dbh is comparable.

It is likely that the importance of landform position and soil association in explaining the abundance of oak seedlings was due to strong relationships between these variables and the availability of light and soil moisture (Geiger 1966). As a rule, northern aspects (NW, N, and NE) are typically more moist and cool than southern aspects (SW, S, and SE) due to lower amounts of incoming radiation (Barnes and others 1998).

The disparity between the patterns of abundance in small and large seedlings across soil associations and landform positions suggests that the relative importance of factors influencing survival may change with seedling development. Greater mean number of small oak seedlings on the northwest/mid-slope landform positions suggests that the moist and cool conditions associated with northwest-facing sites may be more important than light for germination and early survival. Greater number of large oak seedlings on the drier and warmer southeastern/mid-slope positions suggests that conditions associated with this slope position may be the most favorable for recruitment into larger seedling size classes. This pattern supports the assertion that oaks are better adapted to droughty conditions than many other hardwoods, and therefore may compete more successfully with other hardwood species on the drier, poorer sites (Thor and others 1969, Abrams 1990).

The contrasting patterns observed in small and large seedlings suggest that the importance of soil moisture may decrease relative to the importance of light as seedlings develop into the larger size classes. This is difficult to determine definitively from our data, however, because the sample size of larger size class seedlings was very low and the difference in mean abundance of large seedlings between the northwest/mid-slope position and southeast/ mid-slope position was slight.

The importance of seed source to oak regeneration is illustrated by the continual occurrence of the importance value of mature oaks of the same species as regenerating oak seedlings in the smaller size class models. In the analyses of the larger oak seedling individual size classes (results not presented), the importance value of seed sources tended to drop out of the regression models, whereas the potential influence of competitors in the overstory and understory tended to become significant.

Positive relationships between the abundance of oak seedlings and abundance of potential competitors such as beech, blackgum, and pine may have resulted from an affinity of these species for the same site conditions as the oak seedlings. The positive relationship between the number of black oak seedlings and pine, for example, may be explained by a similar tolerance of dry sites shared by pine and black oak (Ashton and Berlyn 1994). Positive relationships between oak seedling abundance and canopy depth, and negative relationships between oak seedling abundance and PAR were unexpected. These relationships may have occurred due to the large proportion of total oak reproduction comprised of oak seedlings that were in the small size classes and relatively shade tolerant (particularly seedlings in the white oak group), and the narrow ranges of PAR and canopy cover present on the study sites.

The negative relationships between oak seedling abundance and the abundance of potential competitors such as hickory, sassafras, dogwood, maple, and mature oaks of other species support the current hypothesis that oak reproduction decreases with increasing abundance of competitors (Lorimer 1993, Hodges and Gardiner 1993). Red maple is replacing oaks in many oak-dominated forests (Lorimer 1984) and dogwood and sassafras are ubiquitous throughout the Ridge and Valley Province. The future of dogwood populations is uncertain, however, due to the spread of dogwood anthracnose throughout the region.

CONCLUSION

While seedling density of smaller size class seedlings is above what has been previously reported, the larger size class seedlings and saplings are well below previous estimates for the Southern Appalachians (Cook and others 1998). For example, the low numbers of white, chestnut, black, and northern red oak seedlings in the larger size classes and the complete lack of scarlet/southern red oak seedlings 151 cm tall up to 8 cm dbh indicate that recapture of these sites by oak following future harvests is questionable. Oak stump sprouts would contribute to a portion of total regeneration, but pre-harvest establishment cuts or other treatments may be necessary to stimulate adequate amounts of larger oak advanced reproduction.

The concentration of large oak seedlings on the southeastern/mid-slope positions tends to support the hypothesis that oaks are more competitive on drier, poorer soils, warmer aspects, and upper slope positions where competitors are limited by growing conditions (Weitzman and Trimble 1957, Abrams 1990, Barnes and Van Lear 1998). Patterns in small seedlings, however, do not support this hypothesis. The hypothesis that oak regeneration should increase with the presence of fewer numbers of potential competitors was not supported by relationships between the abundance of oak seedlings and some potential competitors, PAR, and canopy structure. The negative relationships indicated between the number of oak seedlings and hickory, dogwood, sassafras, and maple are consistent with this hypothesis, and support previous assertions that control of these species may be necessary to facilitate oak regeneration in the region.

Whether the disparity in the abundance of large and small oak seedlings between soil associations and landform positions is due to a change in the relative importance of moisture and light as seedling development proceeds, or not, it appears that the abundance of small seedlings may be a poor indicator of the most favorable sites for obtaining larger oak seedlings and saplings in the future. Despite large differences between species in the number of seedlings less than 25 cm tall, the comparable numbers of seedlings in the larger size classes indicate that recruitment into larger seedling size classes is reasonably comparable for all species sampled except scarlet and southern red oak.

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LARGE WOODY DEBRIS IN A SECOND-GROWTH CENTRAL APPALACHIAN HARDWOOD STAND: VOLUME, COMPOSITION, AND DYNAMICS

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ABSTRACT.—We estimated the volume of large woody debris in a second-growth stand and evaluated the importance of periodic windstorms as disturbances in creating large woody debris. This research was conducted on a reference watershed (Watershed 4) on the Fernow Experimental Forest in West Virginia. The 38-ha stand on Watershed 4 was clearcut around 1911 and has been undisturbed by management activities since that time. Down dead wood (DDW) was sampled in 1999 and identified by species and decay class. The contributions from two windstorms, in 1993 and 1998, were quantified at the same time. Total volume of DDW in Watershed 4 was 69.7 m^3 ha^{-1}, a value equal to or greater than those reported for old growth in central/eastern hardwood forests. The most frequent and the largest volume of DDW was chestnut, followed by sugar maple. Standing dead trees (snags) provided another 41.4 m^3 ha^{-1} of large woody debris. Of the DDW contributed by the windstorms, the largest volume was American beech. The two windstorms were found to contribute approximately 3.5 m^3 ha^{-1} of DDW each. Using these data and other stand characteristics, the stand on Watershed 4 was compared with old growth stands, and implications for management discussed.

Coarse woody debris (CWD), which includes both down dead wood (DDW) and standing dead trees (snags), plays many important ecological functions in forests: providing wildlife habitat (Loeb 1996, Hagan and Grove 1999, Loeb 1999, Menzel and others 1999), creating niches for regeneration of various plant species, and cycling nutrients (Boddy 1983, Harmon and others 1986, Fisk and others 2002), among others. Despite the importance of CWD, the quantity and dynamics have been documented for only a few forest ecosystems in the U.S., and most often in old growth forests (Spies and others 1988, Tyrrell and Crow 1994, Jenkins and Parker 1997). Only recently has the effect of forest management on CWD received attention in the Appalachians (Hardt and Swank 1997). CWD dynamics in central Appalachian hardwood forests, most of which are second-growth, naturally regenerated stands, are not well documented.

In forests throughout the central Appalachians, numerous root mounds of varying ages are obvious. The source of these mounds, some of which are quite old, is not well known, but many mounds are believed to be due to infrequent but severe local windstorms. Romme and Martin (1982) characterized gap production in an old-growth mixed mesophytic forest in Kentucky and identified local, high intensity windstorms as a potentially significant contributor to CWD inputs. However, inputs of CWD were not quantified. For this paper, we measured density, volume, and composition of CWD in a 90-year-old second-growth hardwood stand in the central Appalachians of West Virginia and estimated the inputs of DDW from two localized severe windstorms. We also discuss stand characteristics in term of management implications.

SITE DESCRIPTION AND METHODS

Watershed 4 (38.73 ha) is a reference watershed with an ESE aspect located on the Fernow Experimental Forest, Tucker County, WV (latitude 39° 04′ N, longitude 79° 41′ W). Topography is rugged and elevations ranges from 737 to 853 m on Watershed 4, with slopes ranging from 10 to 60 percent. The old-growth stand was harvested around 1911, leaving some residual trees that were inaccessible or of lower

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value, such as sugar maple, black birch, beech, and hickory. The stand regenerated naturally. Some standing dead American chestnut trees were removed in the 1930s and 1940s (mostly pole-sized). Otherwise there has been no harvesting in this stand since the original harvest. This watershed has been well characterized as part of long-term hydrometeorological studies on the Fernow watershed (Adams and others 1994). Red oak site index 50 ranges from 64 to 81.

In March 1993, a significant winter storm with high winds moved through the area. Within the Monongahela National Forest, an estimated 116 ha of timber was blown over and required salvage logging (Linda White, USDA Forest Service, personal communication). Similar damage was estimated to have occurred on private land adjacent to the National Forest. Significant blowdown was observed on Watershed 4 and another nearby research watershed. Windblown trees on Watershed 4 were cut off at the stump for safety reasons. Where the logs lay across the road at the top of the watershed, they were removed. Otherwise, logs remained where they fell. Another similar storm in April 1998 also resulted in a considerable amount of blowdown and partially uprooted trees on Watershed 4 and the surrounding area. Once again, trees only were removed from Watershed 4 from roads or if they presented a safety hazard.

During September 1998, DDW was measured on Watershed 4 using the transect intersect method (Harmon and Sexton 1996). To achieve a 20 percent precision in our samples (per Martin 1976, Pickford and Hazard 1978), thirty 100-m transects were randomly located within Watershed 4. Along each transect, all DDW larger than 10.2 cm diameter and longer than 1 m was tallied. Species, small and large diameters, length, and wood decay class were recorded for each intersection. Decay classes ranged from 1 to 3, with decay class 1 being the least decayed and decay class 3 being the most advanced stage of decay. (See Adams and Owens [2001] for a more detailed definition of these decay classes.)

Windstorm origin (1993 or 1998) also was noted for every piece of DDW encountered when possible. Other specific wind events could not be identified and all other DDW was classified as "pre-1993". Volume was calculated using the formulae of Van Wagner (1968) and summed for the area. The number of intersections also was summed. The Importance Value was calculated as the sum of the relative volume and relative number of intersections, divided by 2. Standing dead wood (snags) and overstory characteristics also were measured as part of a 100 percent cruise in 2001. Decay class was not assessed for standing dead wood.

RESULTS

Twenty-two species of trees were identified as DDW (table 1). The most commonly encountered species of DDW was American chestnut (for scientific names see table 1). The second most abundant species, based on importance values, was sugar maple, followed by northern red oak.

American chestnut also represented the largest volume of DDW (fig. 1). Sugar maple and northern red oak made up the second and third largest volume of DDW, followed by black locust and American beech. Black cherry also was present in significant amounts. The largest diameter recorded for DDW was a 96.5 cm (38 inches) sugar maple. Total DDW volume for the stand was estimated at 69.7 m^3 ha^{-1}. Overall, 27.9 percent of the DDW was in decay class 1 (least decayed), 40.8 percent in decay class 2, and 31.2 percent in decay class 3 (table 2).

There was an average of 48 standing dead snags per hectare, with a basal area of 5.2 m^2 ha^{-1} and a volume of 41.4 m^3 ha^{-1}. Snags represented 37 percent of total CWD (DDW and standing dead combined), and approximately 16 percent of the standing live volume. Northern red oak was the dominant snag species (of 24 species), followed by sugar maple, black cherry, and black locust. Sixty-nine percent of the standing dead was less than 35.6 cm in diameter and 50 percent was 20.5 cm or less in diameter. The snag with the largest diameter at breast height (dbh) was a 107 cm sugar maple.

Twenty-five species of live overstory trees were recorded. The dominant overstory species on Watershed 4 was northern red oak. Red oak trees were relatively abundant and larger in volume. On the other hand, sugar maple and red maple were abundant but generally smaller in diameter than the northern red oak. Black locust, which is ranked high in the importance values for DDW, is ranked fairly low in overstory importance. The relative importance values for DDW, standing dead, and overstory trees are shown in figure 2. Stand density (stems >10 cm dbh) was 292 trees ha^{-1} with 9.7 trees ha^{-1} having diameters greater than 75 cm dbh. Overstory basal area was 33.3 m^2 ha^{-1}, and estimated volume of 263.8 m^3 ha^{-1}. Stand summary values are shown in table 3.

Table 1.—*Scientific and common names, importance values, and wood decay resistance of tree species represented by down dead wood (DDW) on Fernow Experimental Forest, WV*

Tree Species	Common Name	Relative Importance	Decay resistance[1]
Castanea dentata (Marsh.)Borkh.	American chestnut	.155	High
Acer saccharum Marsh.	Sugar maple	.133	Low
Quercus rubra L.	Northern red oak	.113	Low
Robinia pseudoacacia L.	Black locust	.075	High
Fagus grandifolia Ehrh.	American beech	.074	Low
Prunus serotina Ehrh.	Black cherry	.070	High
Quercus prinus L.	Chestnut oak	.062	High
Betula lenta L.	Sweet birch	.055	Low
Magnolia acuminata L.	Cucumber magnolia	.055	Low
Sassafras albidum (Nutt.) Ness	Sassafras	.051	High
Acer rubrum L.	Red maple	.045	Low
Quercus alba L.	White oak	.042	High
Fraxinus americana L.	White ash	.023	Low
Liriodendron tulipifera L.	Yellow-poplar	.017	Low
Carya cordiformis (Wangehn.)K.Koch	Bitternut hickory	.005	Low
Carya ovata (Mill.) K.Koch	Shagbark hickory	.005	Low
Betula lutea L.	Yellow birch	.004	Low
Oxydendrum arboreum (L.) DC.	Sourwood	.004	Not estimated
Magnolia fraseri Walt.	Fraser magnolia	.003	Low
Tilia americana L.	American basswood	.002	Low
Nyssa sylvatica Marsh.	Black gum	.002	Not estimated
Cornus florida L.	Flowering dogwood	.002	Not estimated

[1] **Based on heartwood decay groupings in Wood Handbook (Forest Products Laboratory 1987).**

In the 1993 and 1998 storms, 6 and 7 species of trees, respectively, were blown over in each storm, with American beech being the most frequent species of DDW created (fig. 3). The two windstorms contributed approximately the same amount of coarse woody debris, about 3 to 3.5 m^3 ha^{-1}. Input from these two storms amounted to approximately 9 percent of the volume of DDW measured on the watershed. Most (88.5 percent) of the dead wood from trees that fell in 1998 was in decay class 1.

More than two-thirds of the trees that fell in 1993 were in decay class 2 (68.5 percent) and the rest (31.5 percent) in decay class 1. None of the trees that fell in 1993 or 1998 were in decay class 3. The more common tree species that fell in 1993 included American beech, sugar maple,

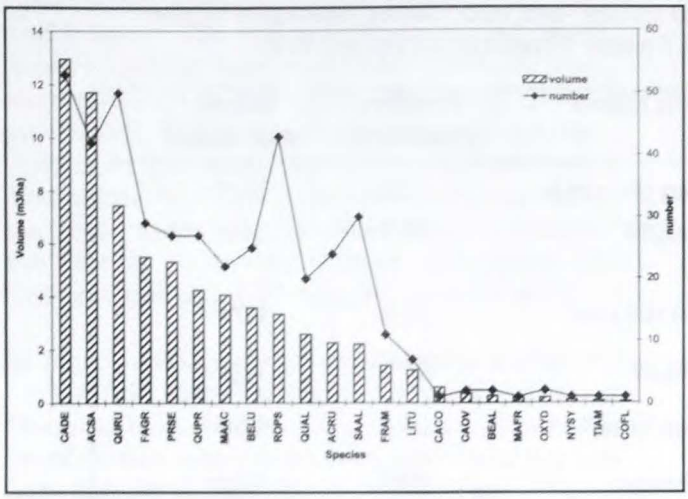

Figure 1.—*Volume and number of down dead wood (DDW) on Watershed 4, Fernow Experimental Forest, WV, by tree species. Species is indicated by the first two letters of their genus and species, except for Oxydendrum (OXYD). See table 1 for genus and species names.*

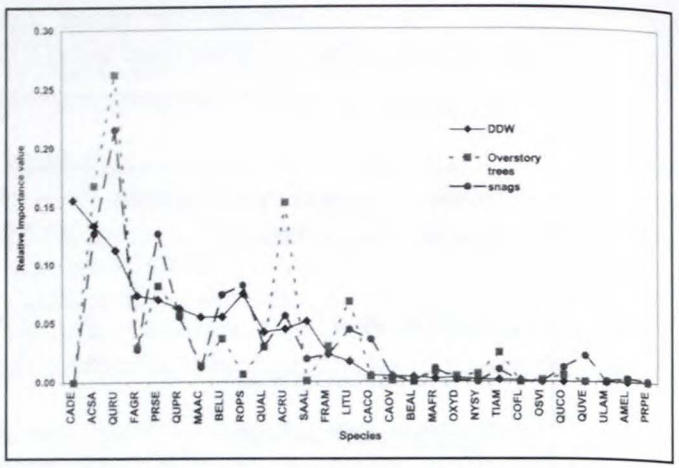

Figure 2.—*Comparison of Importance Values (see Methods for calculation) of down dead wood (DDW), standing dead snags, and live overstory trees, by species, Watershed 4, Fernow Experimental Forest, WV. Species is indicated by the first two letters of their genus and species, except for Oxydendrum (OXDY) and Amelanchier (AMEL). See table 1 for genus and species names.*

Table 2.—*Frequency (percent) of DDW by decay class and time of fall, Watershed 4, Fernow Experimental Forest, WV*

Data set/year	Decay class 1	Decay class 2	Decay class 3
All DDW/years	27.9	40.8	31.2
1998 Storm	88.5	11.5	0
1993 Storm	31.5	68.5	0
Pre-1993 DDW	24.8	40.7	34.41

Table 3.—*Summary of live and dead wood characteristics, Watershed 4, Fernow Experimental Forest, WV*

	Volume (m^3 ha^{-1})	Density (n ha^{-1})	Richness	Evenness	Shannon-Wiener Index (H')
Canopy	263.8	292	25	0.716	2.304
Standing dead	41.4	48	25	0.798	2.568
DDW	69.7	11	22	0.843	2.605

and yellow poplar, while the primary species that fell in 1998 were American beech, red maple, and cucumber magnolia. Species that were found only in the older DDW (did not fall in 1993 or 1998) included American chestnut, sweet birch, hickory, chestnut oak, sassafras, and black locust (fig. 4).

DISCUSSION

CWD composition is clearly a function of several interacting factors: presence in the overstory, disturbance, tree size, and longevity/resistance to decay. For example, distribution of DDW by species generally reflected living stem species composition, with two exceptions: American chestnut and black locust. Both are important DDW, but not in the overstory. Chestnut no longer exists as a canopy species, therefore its presence reflects its previous abundance and great resistance to decay. Black locust is commonly represented among DDW and snags (fig. 2), but much less so in the overstory. Black locust is a shade-intolerant early successional species and short-lived (approx. 40-50 years; Strode 1977). Therefore, it is perhaps not surprising to see it less important in a 90-year-old overstory. Because of the high decay resistance of black locust wood, however, it remains important for snags and DDW for a longer time than almost all other species.

Although red maple is important in the overstory, it is not particularly so for snags or DDW. American beech has low decay resistance, and its relative importance as DDW is probably due to repeated inputs of large decadent branches or tops, such as in the two windstorms recorded here. All three components of the stand have

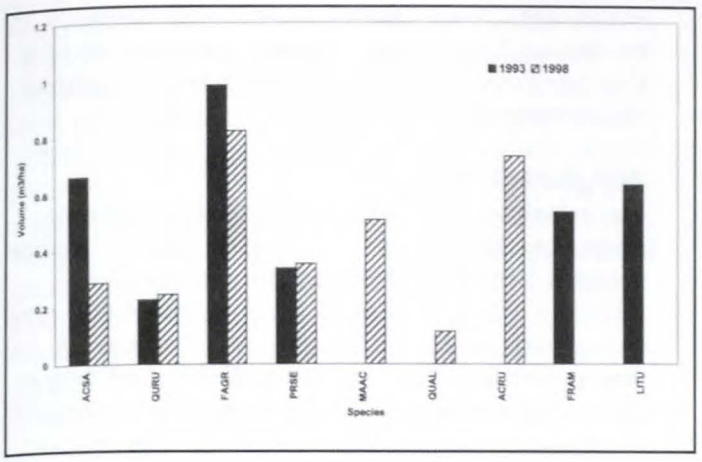

Figure 3.—*Volume of down dead wood delivered from two windstorms, in 1993 and 1998, by tree species. Species is indicated by the first two letters of their genus and species. See table 1 for genus and species names.*

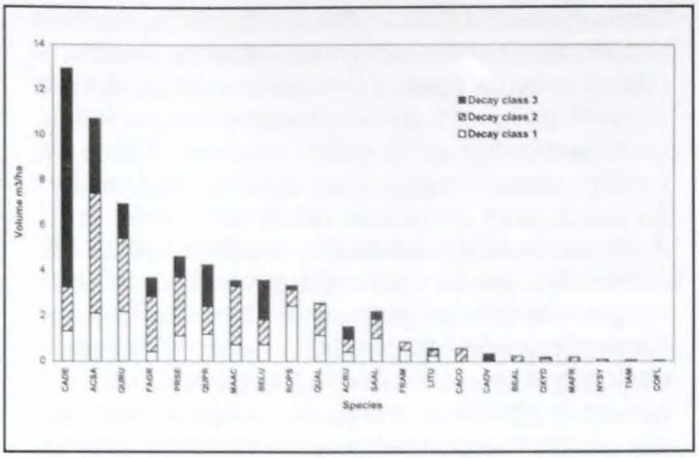

Figure 4.—*Volume of pre-1993 down dead wood (see Methods for definitions) by species and decay class. Species is indicated by the first two letters of their genus and species. See table 1 for genus and species names.*

relatively high species diversity (table 3). In addition to the differences highlighted above, there are a few noncommercial or rare species that are not present as snags or DDW, due to their relative rarity and smaller size.

The two windstorms each contributed about 5 percent of the DDW present at the time of the survey. Although these types of storm are believed to be uncommon, they obviously can contribute significant amounts to the DDW loading in a stand (Spetich and others 1999), and may be an important contributor of DDW to such stands. The distribution among decay classes for these two age groups varies with the age of DDW (the length of time on the ground). However, the two storms did not significantly alter the distribution among classes in total: pre-1993 and total values of DDW distributions among decay classes are approximately the same (table 2). This suggests that these events, while apparently periodic and severe, do not significantly change the longer-term equilibrium of DDW. Hardt and Swank (1997) suggested that near-even distribution of DDW among decay classes provided evidence of regular inputs, as from gap-phase dynamics. The slightly greater volume in decay class 2 would suggest that a pulse of DDW inputs had occurred relatively recently in the past, perhaps as a result of windstorms.

We did not expect that American chestnut would be the dominant form of DDW on Watershed 4. American chestnut previously made up 25 to 50 percent of the overstory in some of these forests (Weitzman 1949), but was nearly wiped out in the 1920s and 1930s by the chestnut blight (*Cryphonectria parasitica*). Although still present in stands today as sprouts (usually less than 15 cm dbh) American chestnut seldom reaches maturity before blight kills the stems. American chestnut wood was widely prized for its durability and it is among the most decay resistant species in the eastern United States (table 1). Most (69 percent) of the American chestnut that was recorded on Watershed 4 was in decay class 3, the most decayed. The largest American chestnut log remaining was 63.5 cm in diameter and the average chestnut log diameter was about 25 cm, suggesting most of the chestnut DDW represents remnants of trees that died at least 60 years ago.

The volume of DDW on Watershed 4 seems high compared to values reported for second growth forests in the Central Hardwood region. Within the Missouri Ozark Forest Ecosystem Project (Shifley and others 1998), DDW volume ranged from 8 to 30 m^3 ha^{-1} for second-growth stands up to 60 years old. DDW volume ranged from 14 to 84 m^3 ha^{-1} in a chronosequence of silvicultural openings in southern Indiana forests (Jenkins and Parker 1997), with the youngest stands (less than 20 years) having greater volumes of DDW than older stands, and the lowest volume (14.71 m^3 ha^{-1}) in the 80 to 100-year-old stands. In a similar assessment in the Southern Appalachians, DDW volume ranged from 22.4 m^3 ha^{-1} to 91.4 m^3 ha^{-1}, with both highest and lowest volumes in the youngest stands (Hardt and Swank 1997).

The volume of DDW on Watershed 4 approaches values reported for old-growth stands. Spetich (1995) noted a total down-wood volume of 74.2 m^3 ha^{-1} in an old-growth forest remnant in southern Indiana. Tyrrell and others (1998) report values ranging from 20.8 to 68.4 m^3 ha^{-1} for mesic and wet-mesic old-growth sites, predominantly in the Midwest, whereas values for old-growth western mesophytic and mixed mesophytic forests range from 66 to 410 m^3 ha^{-1} (Greenberg and others 1997). Some of these mesophytic stands included large American chestnut DDW and snags, particularly those in the southern Appalachians. In an old-growth forest in eastern Kentucky, Muller and Liu (1991) reported a CWD volume of 66.3 m^3 ha^{-1}, of which 11 percent was American chestnut.

Overstory characteristics also fall within the range reported for old-growth forests. The density of snags on Watershed 4 (48.25 trees ha^{-1}) falls within the range published by Tyrrell and others (1998) of 13 to 168 trees per hectare, snag basal area values ranging from 1.3 to 6.8 m^2 ha^{-1}, and snag volumes ranging from 1.8 to 74.8 m^3 ha^{-1}. Values for western and mixed mesophytic forest include 10 to 70 snags per hectare (Greenberg and others 1997). Live stand basal area for Watershed 4 also falls within the range summarized by Tyrrell and others (1998), although stand basal area usually recovers more quickly from disturbance than other stand characteristics (Clebsch and Busing 1989). Martin (1992) suggested at least seven trees per hectare greater than 75 cm dbh is indicative of old growth, while Greenberg and others (1997) report a range from 8.5 to 44.3 trees >75 cm dbh per ha.

On Watershed 4, there are 9.7 trees per ha with a dbh greater than 75 cm. The largest live tree diameter recorded on Watershed 4 was a 122 cm (48 in.) northern red oak, falling within the range (55-127 cm) recorded for black oaks in mesic and wet-mesic northern oak stands (Tyrrell and others 1998). For mixed mesophytic stands, Greenberg and others (1997) list a range of maximum diameters from 104 to 195 cm, but does not include any oak species. Finally, the overstory diversity is high, meeting some of the criteria proposed by Martin (1992) for canopy community composition for old-growth, mixed mesophytic forests: 1) 20 species or more, and 2) evenness of 0.50 or greater. The Shannon-Wiener Index (H') for Watershed 4 is slightly less than the 3.0 proposed by Martin (1992). Although some characteristics, including stand age, do not match all the definitions developed by various scientists, we believe that this second-growth stand has many structural characteristics of old-growth stands.

IMPLICATIONS

Many believe that old-growth forests take several hundred years to develop in parts of the eastern United States and the only way to increase old growth is to leave areas of potential old growth undisturbed for several hundred years. However, the stand on Watershed 4 has developed many of the structural characteristics of an old-growth forest in less than 100 years. Thus, by considering the development of Watershed 4, we may be able to develop some guidelines and management ideas for encouraging old-growth attributes in secondary stands.

Although Watershed 4 is designated as a reference watershed, it is not free from disturbance. Cut in 1911, residual trees of lower value species, or those less accessible, were left behind. These included species like sugar maple, black birch, beech, and hickory, which were not considered merchantable at that time. In a 1933 report on the area that was to become the Fernow Experimental Forest, the forester estimated that "a few hundred feet of beech, birch, and maple" were left after logging in 1911 (Trimble 1977). It was also noted that there were dense stands of American chestnut sprouts, attesting to its importance in the developing stand.

Chestnut blight was undoubtedly a serious disturbance because stands in the area often contained 25 percent by volume of chestnut (Weitzman 1949). The major loss of chestnut in this area occurred in the late 1920s. Loss of such a significant overstory component in such a short time span may have acted to release other species in the young stand, perhaps favoring oak, which perhaps eventually lead to dominance of northern red oak in this stand. Repeated windstorms, such as the two documented in this study, would have created large, infrequent contributions of large woody debris and created larger, multi-tree gaps in the canopy.

Is it possible to use intensive forest management to mimic these processes in Watershed 4, to create "functional old growth," defined here as stands that provide the structure, habitat, and processes found in true old growth, in less than 100 years? True old growth is rare in the central Appalachians and stands are very small and isolated. Although there are as yet no documented old-growth obligate wildlife species in

the region, there are many species such as Virginia northern flying squirrel (*Glaucomys sabrinus*), cerulean warbler (*Dendroica cerulea*), and pileated woodpecker (*Dryocopus pileatus*), which exhibit a preference for old-growth habitat attributes. Old growth is clearly lacking in representation in the central Appalachian landscape. To create desirable old-growth structural attributes such as those needed for the aforementioned wildlife, we propose, using Watershed 4 as a case study, to consider how to develop functional old-growth attributes in 100 years rather than in 300-plus years.

We propose that the following intensive forest management can be used to mimic the milestones mentioned:

1) Very heavy cutting and removal of valuable timber but with scattered older residuals would provide the greatest probability of regenerating a diverse stand of trees, including shade-intolerant species. It will be important to leave only some minimal level of residual stocking (e.g., 2 m^2 ha^{-1} of basal area) in order to ensure the establishment and development of a diverse stand containing a high proportion of intolerant species (Miller and Kochenderfer 1998). Such a cut may prove to be particularly advantageous in regenerating stands that have suffered repeated high-grading, as has been commonly practiced in the Appalachians. Note that in many locations in the eastern United States, controlling white-tailed deer (*Odocoileus virginianus*) herbivory might be necessary, however, to ensure successful diverse regeneration.
2) At approximately 10 years of age, conduct a precommercial crop tree release of intolerants, and particularly where present, northern red oak. This will be necessary to ensure its development among faster growing intolerant species. It is important to remove sufficient stems to provide release of intolerant co-dominant/intermediate trees.
3) At approximately 20 years of age, conduct another precommercial crop tree release of northern red oak and other selected crop trees. Leaving the wood on the ground would provide another source of CWD.
4) Repeated thinning or patch clearcuts would provide additional CWD and release intolerants.

Another approach might be to create small clearcuts to regenerate intolerant species in order to maintain diversity, with designated portions of stands in sensitive areas (e.g., riparian zones) to be left completely unmanaged. In many cases, cutting and leaving all fiberwood and other low value trees on site would have a minimal economical impact on harvesting revenues (J.N. Kochenderfer, unpublished data).

This proposal is based on a single case study, and thus should not be inferred for wide-spread application. Rather, we urge careful consideration of these ideas and site-specific evaluation. Further research and evaluation is clearly called for. It is important, however, to begin to consider ways that we can intensively manage our important hardwood forests for a variety of desired future conditions, and to think "outside the box" in doing so.

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RELATING PAST LAND-USE, TOPOGRAPHY, AND FOREST DYNAMICS IN THE ILLINOIS OZARK HILLS

Saskia van de Gevel, Trevor B. Ozier, Charles M. Ruffner, and John W. Groninger[1]

ABSTRACT.—Trail of Tears State Forest is a 5,200 acre tract in the Illinois Ozark Hills and represents one of the largest blocks of contiguous forest in the lower Midwest. A highly dissected terrain with long, narrow ridges that fall away sharply on either side characterizes the area. The forest cover is a mosaic of oak-hickory approaching "old growth" condition interspersed with bottomland hardwoods. Since the onset of post-European settlement in the 1830s, humans through logging, grazing, and row cropping have manipulated this tract. While past land-use and topography clearly have an impact on present forest cover, the exact nature of this relationship is not known. Given these conditions and the existence of a detailed continuous forest inventory, this forest presents a unique opportunity to explore the interrelationship between topography, past land-use, and present forest condition. A continuous forest inventory was started at Trail of Tears State Forest in 1979. In 1999, we resampled 240 plots and added 100 new plots to develop a 20-year forest stand history. In addition to the forest inventory, dendroecological and historical ecology studies have been implemented to elucidate the influence of natural and anthropogenic disturbances on forest structure and composition. Historical disturbance patterns and their impact on present forest condition can now be described. The results of the continuous forest inventory and historic analysis are being used to develop guidelines for future management (Groninger and others 2002).

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GAP CHARACTERISTICS OF SOUTHEASTERN OHIO SECOND-GROWTH FORESTS

David M. Hix and Katherine K. Helfrich[1]

ABSTRACT.—Transect sampling was used to assess the features of 30 gaps encountered in upland oak stands on the Wayne National Forest. Tip-ups caused the most canopy gaps (52 percent), two-thirds of which were small (< 100 m^2). Only on south-facing slopes were more large gaps found than small gaps. Gaps in the youngest age class (1 to 9 years) were significantly larger than older gaps. Twenty-seven percent of the gaps were shaped like an ellipse. Sugar maple was the dominant tree species successfully regenerating in both small and large gaps. These results indicate that gap dynamics will likely result in a species composition shift from oaks to mesic species in the future.

Most second-growth forest stands of the mixed mesophytic forest region (Braun 1950) regenerated under different disturbance regimes than the remaining old-growth stands (Runkle 1996). Both types of stands experience gap dynamics, yet few studies have focused on naturally occurring gaps in second-growth forests of the Central Hardwood region (e.g., Clinton and others 1993, Wilder and others 1999). A gap is defined by Runkle (1992) as "an area within the forest where the canopy (leaf height of tallest stems) is noticeably lower than in adjacent areas." More studies are also needed that examine gap dynamics on steeply sloping terrain, since these are most often the areas that have been continually forested due to their relative inaccessibility for agriculture (Clinton and others 1994).

Gaps occur in forests for a variety of reasons, e.g., wind damage, individual-tree mortality due to old age. The cause of a gap (or origin type) influences forest microenvironment (i.e., light intensity and available moisture) that in turn affects stand structure, including the composition and distribution of plant species (Putz and others 1983, Battles and others 1996). Origin types are expected to be stand- and region-specific because tree mortality is closely related to various environmental variables and to community characteristics.

The size and shape distributions of gaps are important because they determine both the amount of space and the types of forest floor microsites available for regeneration. Cho and Boerner (1991) found that in moderately well-drained to poorly drained Ohio old-growth forests, most gaps were < 100 m^2. Runkle (1982) also found mostly small (50 to 100 m^2) gaps in old-growth mesic forests in the eastern United States. Gaps on relatively dry, south slopes may be expected to be larger in area (≥ 100 m^2) because vegetation will grow more slowly and canopy openings will not fill as quickly as occurs on mesic slopes (Anderson 1954). Finally, many gap studies estimate area by assuming an elliptical shape for all gaps (Runkle 1982, Lorimer and Frelich 1989, Cho and Boerner 1991). This model can be tested against other calculation methods.

This study focuses on the characteristics of gaps on steep slopes in second-growth upland Central Hardwood forest stands. The major objectives were:

1) to determine the most common types of gap origins,
2) to assess the sizes (areas) and shapes of gaps, and
3) to describe the regeneration of tree species occurring in canopy gaps.

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STUDY AREA

The location for this study was southeastern Ohio, specifically the Marietta Unit of the Wayne National Forest. All areas studied were located in Washington County, which is bordered on the south by the Ohio River. Sampling was limited to forest stands in upper- and mid-slope positions. A prominent feature of the study area is the steep topography. The slope percent was measured on each sample area, and ranged from 25 to 88 percent (mean gradient was 45 percent). The soils of these forest ecosystems are mapped as one of two fine, mesic series: Upshur Hapludalfs (silty clay loam and clay) or Gilpin Hapludults (silt loam) (Lessig and others 1977).

The study area is within the mixed mesophytic forest region (Braun 1950), although the most common overstory trees are upland oaks (Griffith and others 1993). The forest stands investigated in this study had been disturbed one or more times since settlement in the late 18th Century (Gordon 1969). The most frequent types of disturbance have been partial harvesting, grazing, and oil and gas extraction. Suppression of most fires has occurred for at least the past 5 decades. The study area has not yet been invaded by the gypsy moth, and a moderate or higher level of mortality, due to oak decline or other causes, was not observed. Mesophytic species (e.g., sugar maple (*Acer saccharum* Marsh.)) often dominate the understory structural layer of stands in the region. Several studies have indicated that given the expected disturbance regime, these species will likely become a major component of the overstory in many local stands in the future (e.g., Downs and Abrams 1991, Drury 1996).

METHODS

Thirteen line transects were installed in portions of the second-growth forest stands described in the previous section. Starting points were randomly determined in the field, and the total length of transect sampled was approximately 4,000 m. Each transect roughly followed the contour and was oriented parallel to the ridge line. Using Runkle's (1992) line intersect sampling method, we recorded the position along each transect when the line intersected the boundary of a canopy gap. A tree was considered capable of creating or making a gap if it was ≥25 cm in diameter at breast height (DBH). Gap makers were identified (species or genus) whenever possible. Regeneration in gaps was assessed by determining which sapling or small tree would likely fill the gap (Helfrich 1998), i.e., the individual with the greatest potential to reach the canopy in the space left by the gap maker(s). Nomenclature follows Little (1979).

The type of origin for each gap was classified into one of the following five categories: tip-up, basal shear, snag, crown-crash, or geologic. A tip-up occurs when a tree falls and is uprooted, lifting a portion of the soil profile up with its root disk and producing pit- and mound-topography (Peterson and Campbell 1993). When the majority of the canopy opening is concentrated where the top of the tree fell rather than near its original bole position, the gap is considered a crown-crash origin type. The crown-crash origin type is normally associated with a tip-up; however, a much different microenvironment is created (i.e., a large amount of coarse woody debris). Basal shear is a fallen tree that has broken off at or slightly above the base of its bole. A snag is a standing dead tree. Some gaps were caused by a geologic event, i.e., slope movement. The null hypothesis that an equal number of observations would be found in each of the five origin type categories was tested using Chi-square analysis, following the procedure described by Brown and Downhower (1988).

Canopy gap areas were measured in the field according to Runkle's (1992) sampling protocol by locating and measuring two perpendicular lines in each gap: one along the longest line visible and one perpendicular to it at the widest section of the gap. Canopy gap area was calculated using three different methods. First, assuming gaps are ellipsoidal in shape, gap areas (m^2) were determined using the following equation: (π*[length*width]/4).

Second, in the field, the perpendicular lines measured in each gap were transferred to graph paper. From these scaled lines, oriented sketches of each canopy gap were drawn. Canopy gap area was estimated by counting the graph paper squares enclosed in each gap drawing and multiplying that number by the representative area of each square (0.9 x 0.9 m).

The third technique used to estimate canopy gap area consisted of fitting the most representative geometric shape(s) (e.g., a parabola in combination with half of an ellipse) over each gap sketch and counting the squares within the shape(s) area. Using one-way analyses of variance (ANOVAs), no significant differences were found among the three canopy gap area

calculation techniques for either the sample of small gaps (< 100 m^2) or of large gaps (≥ 100 m^2) encountered in this study (small: $F_{2,59} = 0.77$, $P = 0.469$; large: $F_{2,30} = 1.47$, $P = 0.247$). Given the above, the values of canopy gap area used in the analyses were determined by averaging the results from the three area calculation methods.

Gap areas were analyzed using non-parametric Kruskal-Wallis tests (Minitab, Inc., Version 11.0). The Kruskal-Wallis test was used, given the large variances of the populations, to compare canopy gap areas among: four aspect categories (north, east, south, and west), three origin categories (tip-up, basal shear, and other), and three age classes (1 to 9 yrs, 10 to 20 yrs, and 21+ yrs).

Gap age was defined as the number of winters a gap has existed (Runkle 1979). To determine gap ages, we measured the radial growth response in overstory trees adjacent to canopy openings. Radial increment cores were taken from two mature trees on the canopy gap border for each gap sampled. The cores were extracted at 1.37 m above the ground. By examining each core and recording the annual radial increments, we detected major releases from competition (approximately 70 to 90 percent growth increase sustained for approximately 10 to 15 years since the previous 15 years (Lorimer and Frelich 1989). The estimated age for each gap was determined by the most recent major release.

RESULTS

Origin of Gaps

The 30 gaps encountered in this study originated following a variety of events (fig. 1). Each gap was formed following the demise of one or more canopy trees, i.e., more than one tree contributed to gap formation in 36.7 percent of the observations. The number of observations in four of the five origin type categories varied significantly from the expected (equal) number ($x^2_{4,46} = 32.7$, $P < 0.0001$). Tip-ups occurred more often than expected (52.2 percent of the total observations) and crown-crash, snag and geologic origins occurred less often than expected (each of these three categories contained 8.7 percent). Basal shear occurred as frequently as expected (21.7 percent of the total observations).

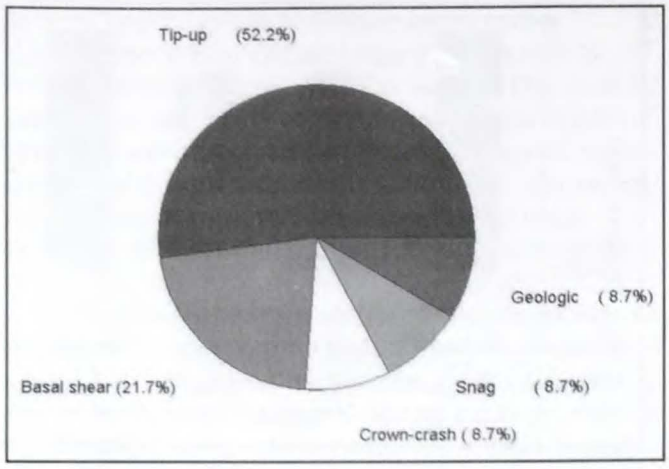

Figure 1.—*Percentages of five origin types for the 30 gaps encountered in second-growth forests of southeastern Ohio.*

Size of Gaps

Small canopy gaps (< 100 m^2) comprised 66.7 percent of the sample and ranged in area from 6.7 to 82.2 m^2. Large canopy gaps (≥ 100 m^2) ranged in area from 101.9 to 291.6 m^2. Overall, the mean canopy gap area was 86.4 m^2 (sd = 58.3 m^2).

Tip-ups tended to create the largest canopy gap areas (table 1); however, there was no significant difference at the 5 percent level in mean canopy gap area among origin types ($H_{2,29} = 5.61$, $P = 0.061$). Mean canopy gap areas also did not significantly differ among slope aspects ($H_{3,28} = 4.65$, $P = 0.200$). Only on the exposed, south-facing slopes were more large gaps found than small gaps (fig. 2).

Gap area was significantly different ($H_{2,27} = 6.77$, $P = 0.034$) among the three age classes, as expected (table 1). Gaps aged 1 to 9 years were

Table 1.—*Means and standard deviations (SDs) of canopy gap areas (m^2) by origin, gap age, and aspect*

	ORIGIN			GAP AGE			ASPECT			
	Tip-up shear	Basal	Other	1-9 years	10-20 years	21+ years	North	East	South	West
Mean	135.9	42.6	49.7	163.2	37.5	95.6	47.0	72.0	137.4	46.7
SD	99.5	21.0	28.6	118.2	16.9	82.0	34.1	39.1	107.1	31.8

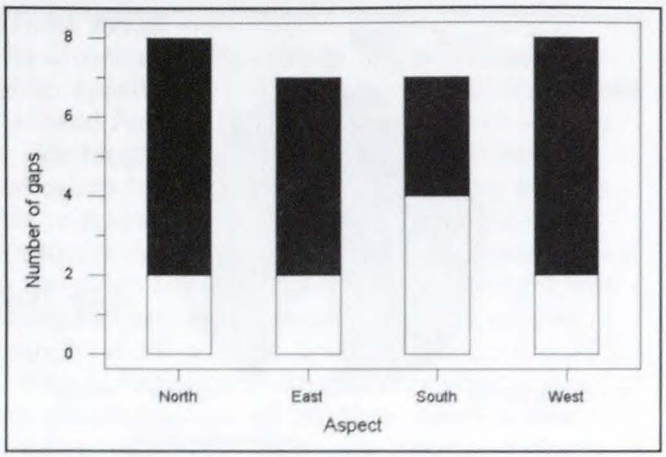

Figure 2.–*The numbers of small (shaded portion of each bar) and of large (≥ 100 m²) gaps found on each of the four slope aspect classes.*

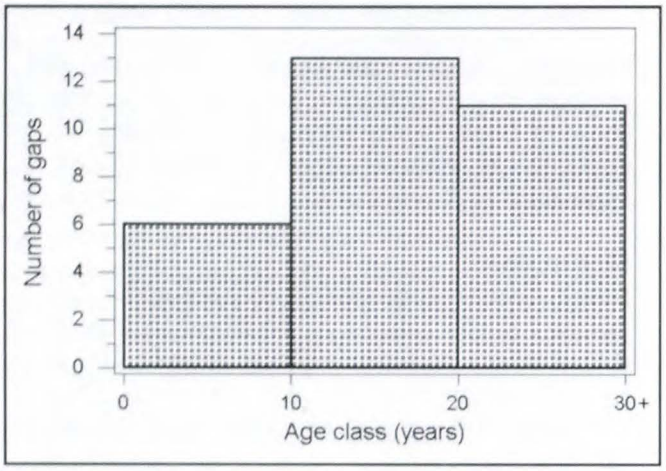

Figure 3.–*The number of gaps in each of three age classes.*

larger than the other age classes in mean canopy gap area. Gaps aged 10 to 20 years were the smallest. Most gaps (63.3 percent) were ≤ 20-years-old (fig. 3).

Shape of Gaps

Most gaps were not shaped like a single ellipse, however, 26.7 percent were judged to most closely resemble single ellipses. The shapes of 43.3 percent of the total gaps were observed to contain an entire or partial ellipse, along with another geometric figure, e.g., circle. Half of the gaps were described as having more than one type of shape.

Gap Maker Trees

Oaks comprised the majority of gap makers in this study. Twenty-five of the 39 gap maker trees were identified: 36.0 percent were northern red oaks (*Quercus rubra* L.), 20.0 percent were chestnut oaks (*Q. prinus* L.), 12.0 percent

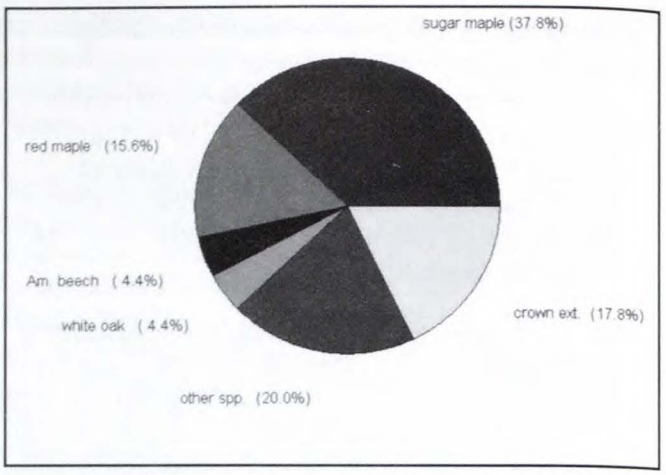

Figure 4.–*Percentages of each species or of lateral crown extension filling the 30 gaps.*

were white oaks (*Q. alba* L.), 8.0 percent were hickories (*Carya* spp. Nutt.), 8.0 percent were maples (*Acer* spp. L.), and 8.0 percent were American basswoods (*Tilia americana* L.). The remainder of the gap maker trees were yellow-poplars (*Liriodendron tulipifera* L.) and elms (*Ulmus* spp. L.). Each of these last two species made up 4.0 percent of the gap maker trees.

Regeneration in the Gaps

Sugar maple saplings and small trees were the most common mode by which gaps were filling (fig.4). Red maple (*Acer rubrum* L.) was the second most common species; 15.6 percent of the gap fillers were this species. In 17.8 percent of the gaps, lateral crown extension was the dominant process of gap closure. Other species that had expressed dominance in 4.4 percent or less of the gaps were American beech (*Fagus grandifolia* Ehrh.) and white oak.

The regeneration in both small and large gaps was primarily sugar and red maple (table 2). In large gaps, these two tree species were filling 76.0 percent of the canopy openings. American beech was the dominant species in 10.0 percent of the small gaps, but none of the large gaps. Species dominating 6.0 percent of the large gaps, but none of the small gaps, were slippery elm (*Ulmus rubra* Muhl.), Ohio buckeye (*Aesculus glabra* Willd.), and hackberry (*Celtis occidentalis* L.).

DISCUSSION

Wind damage is associated with tip-ups (Peterson and Pickett 1991) and tip-ups are associated with large gaps (Whitmore 1975). In this study, the majority of the gaps were created by tip-ups; however, most of the gaps were small in area. The relatively high frequency of

Table 2.—*The percentages of each tree species filling both small and large gaps*

	Small Gaps	Large Gaps
Acer saccharum	45	57
A. rubrum	20	19
Aesculus glabra	0	6
Carya tomentosa	5	0
Celtis occidentalis	0	6
Fagus grandifolia	10	0
Liriodendron tulipifera	5	0
Quercus alba	5	6
Q. prinus	5	0
Q. rubra	5	0
Ulmus rubra	0	6

tip-ups encountered compared with the frequency of tip-ups reported in other studies (e.g., Krasny and Whitmore 1991) may be due to differences in the stands' species compositions. Oaks comprised the majority of the gap makers in this study, and, because of their wood properties, trees of this genus may be more susceptible to windthrow than those of other tree species.

In a study by Putz and others (1983) of a Panama tropical rainforest, tree species with short, stocky boles and dense, strong wood (similar to oaks in second-growth forests) uprooted relatively more often than species not generally possessing those characteristics. Additionally, Putz and others (1983) found that frequency of origin type did not significantly differ with changes in degree of slope, so species composition likewise may have had more influence on the high frequency of tip-ups in this study than did the steepness of the topography.

In contrast, Runkle (1982) found the origin of most gaps in eastern mesic old-growth forests was basal shears, and only 19 percent of origins were tip-ups. Oaks were not the dominant species in the stands studied by Runkle (1982). Similarly, Whitmore (1975) found that basal shear in tropical rainforests created smaller gaps than tip-ups. The above trends are intuitive; tip-ups involve an entire tree falling at once, while basal shear involves only some portion of the tree falling and snags gradually deteriorate over time (Krasny and Whitmore 1991).

Most gaps (67 percent) were small (< 100 m^2). Small gaps were also numerous in a study of a second-growth cove forest in the Great Smoky Mountains (Clebsch and Busing 1989). Part of the reason for the abundance of small gaps and lack of large gaps is the absence of major disturbances that cause large gaps in southeastern Ohio (Goebel and Hix 1996). The lack of large gaps can also be attributed to the second-growth forest structure—basal area is not concentrated in a few large trees capable of creating larger gaps like in the tropics where canopy height is also relatively higher (Whitmore 1975, McCarthy and others 1987). Among the four aspect classes, only on exposed, south-facing slopes were more large gaps found than small gaps. The prevailing winds along with the soil water limitations of south-facing aspects may cause gaps to fill slowly, with the result being the presence of relatively more large gaps on these slopes.

As a gap ages, lateral crown extension of the surrounding trees fills it and the size of the gap becomes smaller (Runkle and Yetter 1987). Therefore, it was expected that the older gaps would be smaller than the younger gaps. Gaps 1-9 years old were the largest, but the 10-20 year age class had the smallest areas instead of the oldest age class (≥ 21 years). One reason for this may be that gap colonization occurs "non-randomly" like origin (Whitmore 1975), and the rate of colonization is fastest between 10-20 years after a gap is created. Most gaps in Runkle's (1982) study were less than 15 years old, while the majority of the gaps we encountered were less than 20 years of age.

Different methods of gap area calculation have been compared within and among various studies (e.g., Runkle 1985, Battles and others 1996). Results have indicated that gap area may be overestimated by assuming an elliptical shape (Battles and others 1996). The majority of gaps we encountered were not single ellipses, and half of the gaps consisted of multiple shapes. In this study, assuming elliptical shapes for all gaps did not significantly overestimate area when compared with our two graphical approaches to gap area estimation. If a quick gap area calculation is desired, an ellipse is the most obvious choice as a model.

The species composition of the gap maker trees closely reflects the composition of forest stands of the study area. Fifty-six percent of the living trees in second-growth forests of Washington County, Ohio, are oaks and hickories (Keller and Hix 1999). In comparison, 76 percent of the identified gap makers were the same species. The regeneration successfully established in both small and large gaps indicates a shift in species composition is very likely to occur in the future.

Although 18 percent of the gaps were closing due to lateral crown expansion, sugar maple and red maple were filling 53 percent of the gaps.

Tree species richness was greater in small gaps than in large gaps (eight versus six regeneration species). Only white oak was found to be successful in both small and large gaps. The conclusions of others (e.g., Drury 1996, Runkle 1996) concerning the successional trajectory over time of similar stands in the region are supported by this study: as canopy gaps naturally occur, oaks will be replaced by mesic species such as sugar maple.

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REGENERATING MIXED OAK STANDS IN PENNSYLVANIA: A QUARTER-CENTURY RETROSPECTIVE

P.J. Gould, K.C. Steiner, J.C. Finley, and M.E. McDill[1]

ABSTRACT.—The outcomes of regeneration treatments in 90 oak-dominated stands in Pennsylvania are examined 20 to 33 years after treatment. Approximately one-quarter of the stands failed to reach 50 percent stocking after at least 20 years, but most stands regenerated successfully. Red maple is the most frequently observed species in the regenerated stands, followed by oak species and sweet birch. The relative basal area of oak decreased from pre-harvest levels while the relative basal area of sweet birch and red maple increased substantially. Oaks and red maple regenerated from both seeds and sprouts, while sweet birch regenerated primarily from seeds. Oaks were generally larger than their competitors and may become an increasingly important component of the regenerated stands.

Regeneration treatments in oak-dominated stands in the eastern United States often result in stands where oaks play a diminished role (Lorimer 1992). Since the mid 1960s, Pennsylvania State Forest managers have been regenerating mixed-oak stands using even-aged silvicultural prescriptions. To date, a comprehensive evaluation of regeneration outcomes has not been conducted. The objectives of this study are:

1) to document the outcomes of regeneration harvests conducted between 1968 and 1978 in 90 oak-dominated stands on Pennsylvania State Forest lands,
2) to determine whether the relative importance of oaks and other species in the regenerated stands differs from the pre-harvest condition,
3) to asses whether oaks may become dominant in the stands where they occur, and
4) to measure the relative importance of seed-origin and sprout-origin oak and non-oak stems in the regenerated stands.

METHODS

Data were collected on a set of 90 oak-dominated stands at two points in time—before regeneration treatment and 20 to 33 years after treatment. Pre-harvest data for each stand were amassed through an extensive search of operational records collected by the Pennsylvania Department of Conservation and Natural Resources, Bureau of Forestry (and precursor agencies) as part of the state's timber management program. Stands were entered into the analysis if three conditions were met:

1) the results of a pre-harvest stand exam could be found,
2) the pre-harvest stand exam indicated that oak species accounted for at least 50 percent of the pre-harvest basal area, and
3) the regeneration harvest occurred at least 20 years prior to 2001.

Pre-harvest stand exams were used to assess the composition of each stand prior to harvest. A prism plot sample (n = 10) of basal area, tallied by species, was included in each pre-harvest exam. The samples were used to calculate the number of stands in which each recorded species appeared and the frequency of appearance within the set of stands (number of appearances/total number of stands x 100 percent). Pre-harvest relative basal area (species basal area/total stand basal area x 100 percent) was calculated by species for each stand. General descriptions of stand condition were frequently included in the pre-harvest exams and were used as qualitative data.

Follow-up stand exams were used to assess the composition of each stand 20 to 33 years after

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stand initiation. In stands of 30 or more acres, approximately 30 prism plots were taken along five or six equally spaced parallel lines. In smaller stands, at least one prism plot was taken for each acre of stand area, with a minimum of 10 plots. The species, crown class (dominant, codominant, intermediate, or suppressed) and dbh (diameter at breast height –4.5 ft) of each "in" tree was recorded (Smith and others 1997).

The stocking level of each stand was calculated for all observed species and for only oak species (Gingrich 1967). The frequency and relative basal area of each species in the regenerated stand was calculated in the same way as in the pre-harvest stands. The quadratic mean diameter (QMD) and basal area of dominant and codominant trees was calculated by species for each stand. Oak species were considered individually and as a group.

Differences between the pre-harvest and post-harvest relative basal area of oaks and other species were compared using paired t-tests, with each stand serving as the pairing variable. The dominance of oaks in the regenerated stands was evaluated by comparing the basal area and QMD of dominant and codominant oaks to those of common competitors. Analysis of variance (alpha < 0.05) was used for the comparisons with each stand serving as a block to account for variation between stands. Consequently, comparisons between oaks and competitors were restricted to regenerated stands in which the species co-occurred.

The contribution of stump sprouts and seedlings to the dominant and codominant basal area of common species was evaluated using a subset of 25 follow-up exams. The exams, completed in 2000 and 2001, were conducted in the same way as described above, except each "in" tree was recorded as either a seed-origin or sprout-origin stem. Origin determinations were based on clumping of stems, stump remnants, and the shape of the root collar. If there was not any evidence that a single stem originated as a stump sprout, it was classified as a seed-origin stem. Stems classified as seed-origin likely include stems that established before the regeneration treatments and re-sprouted after treatment (i.e., seedling sprouts).

Relative basal areas were calculated as the percentage of each species' total basal area originating from sprouts or seeds. Seed and sprout-origin relative basal areas and QMDs were compared for each species using paired t-tests, with stand as the pairing variable.

RESULTS

The distribution of stocking levels in the regenerated stands is bimodal, with almost one-quarter of stands failing to reach a stocking level of 25 percent and most of the other stands reaching stocking levels in excess of 75 percent (table 1). Heavy pre-harvest mortality, usually due to defoliation by the oak leaf roller (*Archips semiferana*), was noted as having occurred in most poorly stocked stands, but some well-stocked stands also had heavy pre-harvest mortality. Stocking of oak species is heavily weighted towards the < 25 percent class, but 10 percent of stands have an oak stocking level in excess of 75 percent. Stocking was not found to be correlated with stand age ($p = 0.481$). Because of the bimodal distribution of stocking levels, further analysis was limited to stands that reached at least 50 percent stocking. The sample size was reduced from 90 to 65 stands after poorly stocked stands were excluded.

Forty-eight tree species were observed in the regenerated stands. The pre-harvest and post-harvest frequencies and relative basal areas of the most common species are listed in table 2. Red maple (*Acer rubrum* L.) is the most common species, occurring in all stands. Oak species

Table 1.—*Percentage of 90 formerly oak-dominated stands in central Pennsylvania that reached each stocking level 20 to 33 years after stand initiation*

	Stocking Intervals (percent)[1]			
Species	**< 25**	**25 - 50**	**50 - 75**	**> 75**
All species	22.2	5.6	3.3	68.9
Oak species	61.1	24.4	4.4	10.0

[1]Calculated from Gingrich (1967)

Table 2.—*Frequency and mean relative basal area of common species before and after regeneration treatments in 65 formerly oak-dominated stands in Pennsylvania. Stands were sampled before treatment and 20 to 33 years after treatment. Differences are based on individual paired t-tests.*

	Frequency		Relative Basal Area		
Species	Before	After	Before	After	P (Difference)
Red maple	93.8	100.0	11.5	37.1	<0.001
All oaks	100.0	98.5	77.7	30.2	<0.001
Chestnut oak	89.2	83.1	31.6	15.4	<0.001
Sweet birch	29.2	80.0	0.9	11.3	<0.001
N. red oak	89.2	75.4	13.5	5.2	<0.001
Black oak	87.7	64.6	3.2	9.9	<0.001
Scarlet oak	67.7	63.1	7.9	3.5	0.004
Black cherry	6.2	60.0	0.2	4.7	<0.001
white oak	89.2	58.3	14.6	2.7	<0.001
Hickories	44.6	52.3	1.0	2.1	0.004
Black gum	30.8	50.8	0.9	1.2	0.275

occur in all but one stand (98.5 percent of stands), and chestnut oak (*Quercus prinus* L.) is the second most common tree species (83.1 percent), followed by sweet birch (*Betula lenta* L.) (80.0 percent). Northern red oak (*Quercus rubra* L.), black oak (*Quercus velutina* Lam.), scarlet oak (*Quercus coccinea* Muenchh.), white oak (*Quercus alba* L.), and black cherry (*Prunus serotina* Ehrh.) all occur in at least 60 percent of the stands. Hickory species (*Carya* spp.) and black gum (*Nyssa sylvatica* Marsh.) both occur in about one-half of the stands. Yellow-poplar (*Liriodendron tulipifera* L.), a ubiquitous competitor in some parts of the eastern United States, appears in only 26.2 percent of the regenerated stands.

The frequencies observed in the regenerated stands were similar to those observed in the pre-harvest stands with some notable exceptions (table 2). Sweet birch, which occurred in only 29.2 percent of stands before harvest is now present in 80.0 percent of stands and black cherry increased from 6.2 percent to 60.0 percent. The frequency of white oak decreased from 89.2 percent of stands before harvest to 58.3 percent of regenerated stands.

The mean relative basal area of each species in the regenerated stands follows its frequency rank in most cases (table 2). Red maple contributes an average of 37.1 percent to total basal area, while the oaks contribute an average of 30.2 percent. Only red maple, chestnut oak, and sweet birch have mean relative basal areas of 10 percent or more.

The mean relative basal areas of most species in the regenerated stands are significantly different from their pre-harvest levels (table 2). The mean relative basal areas of red maple, sweet birch, and black cherry increased ($p < 0.001$ in all cases). The mean relative basal area of the combined oak species and of each individual oak species decreased, with the exception of black oak, which experienced an increase ($p \leq 0.004$, in all cases).

The QMD and mean basal area of dominant and codominant red maple, sweet birch, and oaks were compared across 47 regenerated stands where the species co-occur (table 3). Oaks have a significantly larger mean QMD (5.4 inches) than red maple and sweet birch (4.4 and 4.8 inches, respectively). The mean QMDs of red maple and sweet birch are not significantly different from one another. Oaks have significantly more basal area than sweet birch

Table 3.—*Quadratic mean diameter and mean basal area of oak species, red maple, and sweet birch in 47 formerly oak-dominated stands in Pennsylvania, USA, where the species co-occur. Means not sharing letters in each column are significantly different at P < 0.05.*

Species	Diameter[1]	Basal Area[2]
Oaks	5.4 a	21.6 a
Red maple	4.4 b	17.7 ab
Sweet birch	4.8 b	7.5 b

[1]Qudatric mean diameter (inches).
[2] in ft^2 $acre^{-1}$

(21.6 $ft^2\ ac^{-1}$ versus 7.5 $ft^2\ ac^{-1}$) while the basal area of red maple (17.7 $ft^2\ ac^{-1}$) does not significantly differ from either oak or sweet birch.

Sprouting played an important role in regenerating the oaks and red maple, but a minor role in regenerating sweet birch (table 4). Sprouts contribute an average of 67.9 percent of the oak basal area in the regenerated stands, a value near the observation made by McIntyre (1936) that 75 percent of second-growth oaks in Pennsylvania were of sprout origin. Sixty percent of the red maple basal area and 2.5 percent of the sweet birch basal area is of sprout-origin.

The relative basal area of sprout-origin oaks is significantly greater than that of seed-origin oaks ($p = 0.006$) while the opposite is true for sweet birch ($p < 0.001$) (table 4). A significant difference was not found between sprout-origin and seed-origin red maple basal area ($p = 0.116$). In most stands, both sprout and seed-origin oaks and red maples were found. Sprout-origin sweet birch stems were found in five stands, always with seed-origin stems. The mean QMDs of sprouts do not differ from the mean QMDs of seedlings for any of the three species.

DISCUSSION

Regeneration treatments in oak-dominated stands resulted in a range of outcomes. On one hand, a sizable percentage of stands failed to reach 50 percent stocking of any tree species. Regeneration failures tended to occur after heavy mortality in the pre-harvest stand, but apparently healthy stands also failed to regenerate. On the other hand, most stands regenerated well and 14.4 percent of stands are already 50 percent or more stocked with oak. In general, an oak component was maintained in most successfully regenerated stands.

Although an oak component was maintained in most stands, oak relative basal decreased from pre-harvest levels. Red maple and sweet birch became more prominent after the regeneration harvests and the two species account for almost 50 percent of the basal area in the regenerated stands, compared to 13.4 percent in the pre-harvest stands. Red maple was present in most of the pre-harvest stands and the regeneration harvests likely released advanced red maple regeneration, as well as stimulating stump sprouting (Lorimer 1984). Both regeneration strategies are reflected in the regenerated stands.

In contrast to red maple, sweet birch invaded stands after the regeneration harvests. Most sweet birch stems are seed-origin and the frequency of occurrence rose from 29.2 percent in the pre-harvest stands to 80.0 percent in the post-harvest stands.

Oaks exhibit signs of dominance in the regenerated stands where they occur. In stands where oaks, red maple, and sweet birch coexist, oaks have significantly larger QMDs. Oak basal area exceeds sweet birch basal area and is not significantly different from red maple basal area. Although stump-sprouts are the most important source of oak basal area in the regenerated stands, the QMDs of sprout and seed-origin stems do not indicate that one source of regeneration is growing faster than the other.

Although oaks appear to play a diminished role in the regenerated stands, the dominance of oak may increase as the stands develop. Oliver (1978) argued for a deterministic pattern of increasing oak dominance with a similar combination of species. Unlike southern Appalachian stands, where yellow-poplar may overtop oaks later in stand development, it seems plausible to expect oak to increase in dominance, especially over red maple (Beck and Hooper 1986). Assuming established oak are out-growing

Table 4.—*Relative basal area and quadratic mean diameter of seed-origin and sprout-origin stems sampled in 25 stands in central Pennsylvania. Differences are based on individual t-tests of samples paired by stand.*

	Relative Basal Area[1]				Quadratic Mean Diameter (in)			
Species	#	Seedlings	Sprouts	P(Difference)	#	Seedlings	Sprout	P (Difference)
Oaks	24	32.1	67.9	0.006	21	7.3	7.0	0.422
Red maple	25	40.0	60.0	0.116	24	5.6	5.2	0.193
Sweet birch	20	97.5	2.5	<0.001	5	5.0	4.8	0.882

[1]Percentage of species basal area from each origin.

potential competitors, the relative basal area of oaks will increase as the stand develops.

CONCLUSIONS

- The regeneration of mixed-oak stands in central Pennsylvania meets with a range of outcomes, from excellent oak regeneration to complete regeneration failure. Most stands that successfully regenerate contain some oak.

- The importance of oak decreased after regeneration harvests, while the importance of sweet birch and red maple increased. Red maple was present in most stands before harvest and regenerated through stump sprouting and seed. Sweet birch was relatively rare in the pre-harvest stands and invaded through the establishment of seedlings.

- Oaks show signs of being superior competitors in the regenerated stands. The relative basal area of oaks may increase as the stands develop.

- Stump sprouting and seedling establishment are both important mechanisms for regenerating oak. Sprout-origin stems are not larger than seed-origin stems after 20 years.

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EFFECT OF PREHARVEST UNDERSTORY TREATMENT AND GROUP OPENING SIZE ON FOUR-YEAR SURVIVAL OF ADVANCE REPRODUCTION IN THE BOSTON MOUNTAINS OF ARKANSAS

Martin A. Spetich and David L. Graney[1]

ABSTRACT.—The purpose of this study was to examine survival of regeneration in small openings. Six forest stands were located in 1990 and three understory treatments were applied to each in early 1991. In each stand, six sizes of openings were created in 1992 ranging in size from 0.025 ac to 0.50 ac. Understory treatments consisted of:

1) no control of competing regeneration,
2) mechanical control of competing regeneration (cut stems), and
3) chemical control of competing regeneration (cut stems sprayed with herbicide).

Only northern red oak appeared to exhibit a response to treatments with increased survival as intensity of understory control increased and as opening size decreased. We attribute these results to the wide fluctuation in seedling populations and the short time interval between opening creation and post-treatment measurement.

Maintaining tree species composition in upland hardwood forests has been problematic. Without historic fire regimes or sufficient light in the understory the trend in species dominance will likely continue to move toward shade tolerant trees (Nigh and others 1985, Dodge 1997). In the Boston Mountains of Arkansas, we periodically find carpets of small, less than 1-foot-tall, first-year oak seedlings. Unfortunately these seedlings rarely survive more than 1 or 2 years on moderate to high site index sites without controlling competition and providing sufficient light (Graney 1999). The difficulty of maintaining species composition is especially challenging when balancing what is ecologically suitable for regeneration of oak and the multiple desires of many publics.

Management with low visual impact has led to consideration of using small group openings in the Boston Mountains of Arkansas. However, the effect of group openings of 0.5 acres or less on species composition in the Boston Mountains has not been examined. The long-term objective of this study is to examine the effectiveness of small group selection openings on survival of white oak (*Quercus alba* L.), northern red oak (*Quercus rubra* L.), black oak (*Quercus velutina* Lam.), black cherry (*Prunus serotina* Ehrh.), white ash (*Fraxinus americana* L.), post oak (*Qurecus stellata* Wangenh.), hickory (*Carya* Nutt.), and black walnut (*Juglans nigra* L.). Our short-term objectives are to:

1) determine survival of regeneration 4 years after opening creation in relation to treatments, and
2) examine the usefulness of logistic regression to examine highly unstable short-term seedling survival in relation to treatments.

METHODS

The Boston Mountains are the highest and most southern member of the Ozark Plateau Physiographic province. They form a band 30 to 40 miles wide and 200 miles long from north-central Arkansas westward into eastern Oklahoma. Elevations range from about 900 ft in the valley bottoms to 2,500 ft at the highest point. The plateau is sharply dissected. Most ridges are flat to gently rolling and are generally less than 0.5 mile wide. Mountainsides consist of alternating steep simple slopes and gently sloping benches.

Soils on mountaintops and slopes usually have shallow to medium depth and are represented by medium-textured members of the Hartsells, Linker, and Enders series (Typic Hapludults).

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They are derived from sandstone or shale residuum, and their productivity is medium to low. In contrast, soils on mountain benches are deep, well-drained members of the Nella and Leesburg series (Typic Paleudults). They developed from sandstone and shale colluvium, and their productivity is medium to high. Rocks in the area are alternating horizontal beds of Pennsylvanian shales and sandstones. Annual precipitation averages 46 to 48 inches, and March, April, and May are the wettest months. Extended summer dry periods are common, and autumn is usually dry. The frost-free period is normally 180 to 200 days long.

Study Description

Six upland hardwood stands were located in 1990. Pretreatment inventories of regeneration were done in late 1990 (July through November). Three understory treatments were applied in early 1991:

1) no control of competing regeneration,
2) mechanical control of all competing stems greater than 1 foot tall and less than or equal to 5.5 inches d.b.h. (cut stems), and
3) chemical control of all competing stems greater than 1 foot tall and less than or equal to 5.5 inches d.b.h. (cut stems sprayed with herbicide).

In 1992 (May through September), six sizes of openings were created: 0.025 ac, 0.05 ac, 0.10 ac, 0.20 ac, 0.35 ac, and 0.50 ac. All trees greater than 5.5 inches d.b.h. were removed in the harvest.

Individual stems of the regeneration to be released were tagged with unique identification numbers to examine survival throughout the study. Due to the expense and time of tagging trees, and following them throughout the study period, only species difficult to regenerate successfully and/or intermediate to intolerant of shade were examined. They included white oak, northern red oak, black oak, black cherry, ash, post oak, hickory, and black walnut. Other species were considered competitors.

Regeneration Measurements

Regeneration was sampled on a series of permanent 0.001-acre (3.72 feet in radius) or 0.002-acre (5.27 feet in radius) plots that were placed in each group opening. Regeneration plots in the 0.025 openings were 0.001-acre while plots in larger openings were 0.002-acre. Each 0.025- and 0.05-acre opening had five regeneration plots in which seedlings were tagged and measured, while larger openings had nine. One plot was placed at the center of the opening while the other plots were placed along the four cardinal directions (North, East, South and West).

Stems less than 1.6 inches d.b.h. were measured in 1990 and again in 1996 on each plot. Measurements included species and ground diameter to the nearest 0.1 inch (measured 1 inch above ground level on the uphill side of the stem). Within each plot, all black oak, northern red oak, and white oak stems were tagged and measured during the first inventory. If less than two stems of an oak species greater than 1 foot tall existed within a plot, then a stem was located and tagged outside the plot, so that at least two stems of each existing oak species was tagged if available within the opening. These additional trees will increase our ability to model long-term survival. In addition, a subset of species including post oak, black cherry, ash, hickory, and black walnut were tagged for each plot. In the subset of species, trees were tagged in proportion to their occurrence and size, with at least two stems of each species being tagged if possible.

Modeling Survival

Logistic regression was used to estimate survival probabilities of tagged regeneration trees. Designed independent variables included three levels of understory weed control and six levels of opening size. The dependent variable is the probability of survival. Although the observed value of the dependent variable is binary (0 or 1) in logistic regression, the resulting probability estimates are continuous and restricted to the interval 0 to 1. To prepare for logistic regression analysis, each tree was assigned either 1 (successful) or 0 (unsuccessful), where mortality of an individual stem by the time of the post-treatment measurement was considered unsuccessful.

For categorical independent variable (understory weed control), standard logistic regression dummy variables were used (Hosmer and Lemeshow 1989). For instance, because there were three weed control treatments, two dummy variables were needed, where the no understory control (U0) was coded 0, 0; manual understory control (U1) was coded 1, 0; and understory control treatment with herbicide (U2) was coded 0, 1.

To evaluate logistic regression model performance, we selected predictors with a p-value of 0.05 or less based on the Chi-square distribution with one degree of freedom. We used the Hosmer-Lemeshow goodness-of-fit statistic (Hosmer and

Lemeshow 1989, p. 140) to test the null hypothesis that the equation described the data. For Hosmer-Lemeshow goodness-of-fit p-values of 0.05 or less (indicating a poor fit of the equation to our data), the null hypothesis was rejected. This means that predictor p-values of 0.05 or less have a different interpretation than the Hosmer-Lemeshow goodness-of-fit p-values of 0.05 or less.

RESULTS AND DISCUSSION

During the post-treatment inventory 4 years after opening creation, 64 percent of the 11,807 tagged seedlings had survived (table 1). White oak, northern red oak, black oak, and black walnut had the highest mortality. The largest proportion of tagged seedlings were represented by white oak at 3,957 seedlings, northern red oak with 2,491 seedlings, and black oak with 1,774 seedlings.

Of the original stems, survival by species was 54 percent white oak, 60 percent northern red oak, 63 percent black oak, 90 percent black cherry, 87 percent ash, 100 percent post oak, 100 percent hickory, and 57 percent black walnut. However, post oak, hickory, and black walnut included only 3, 1, and 19 stems, respectively. Percent survival by understory treatment, species, and opening size does not appear to present any clear consistent trends at this early stage (table 1). We further inspected treatments in relation to survival using logistic regression to determine the usefulness of logistic regression to examine short-term data of this type.

Based on our logistic regression, survival probability of northern red oak increased with increasing intensity of understory control treatment and decreasing opening size. We suspect that the increasing survival with decreasing opening size is due to the large natural fluctuation in young seedling survival and the short time interval between opening creation and post-treatment measurement. The model is:

$$Pnro = 1/(1+EXP(-(0.175-(1.218*O)+U)))$$

where Pnro = the probability of northern red oak survival, O = Opening Size in acres, U = 0 for no understory control, U = 0.458 for mechanical control and U = 0.364 for chemical control.

Predictors for opening size and post-treatment are significant at $\alpha = 0.05$ ($pO < 0.001$, $pU < 0.001$). Based on the Hosmer-Lemeshow goodness-of-fit statistic, differences between estimated probabilities and observed responses are not significant ($p = 0.480$). The model is based on 2,491 observations. Any model describing this early stage of development in small openings should be viewed with skepticism due to the high instability of early reproduction in these systems resulting in large fluctuations in population density. For instance, in the case of northern red oak, 40 percent of the trees had died by the time of the second measurement.

Table 1.—*Percent survival by understory treatment, species and opening size. Numbers in parenthesis indicate number of tagged seedlings prior to treatment.*

Understory Treatment	Species	Opening Size (acres) 0.025	0.050	0.100	0.200	0.350	0.500
None	Ash	100(29)	98(23)	98(35)	85(49)	84(38)	77(56)
Mechanical	Ash	89(19)	98(18)	82(42)	86(33)	84(49)	95(58)
Chemical	Ash	90(11)	94(16)	87(22)	91(20)	82(56)	57(44)
None	Black cherry	89(26)	96(21)	99(50)	92(72)	81(82)	90(77)
Mechanical	Black cherry	91(33)	94(42)	93(47)	97(84)	95(88)	86(102)
Chemical	Black cherry	97(23)	83(38)	85(63)	94(61)	86(74)	76(72)
None	Black oak	47(40)	63(29)	69(148)	68(90)	53(161)	55(151)
Mechanical	Black oak	76(37)	51(27)	69(126)	53(101)	62(243)	54(148)
Chemical	Black oak	76(38)	76(51)	78(76)	54(70)	67(102)	67(136)
None	N. red oak	67(93)	63(68)	66(237)	56(122)	33(149)	61(174)
Mechanical	N. red oak	69(70)	62(124)	73(138)	54(194)	61(274)	49(288)
Chemical	N. red oak	77(40)	60(57)	63(91)	61(65)	62(137)	48(170)
None	White oak	51(77)	47(113)	48(85)	61(163)	49(298)	60(259)
Mechanical	White oak	52(110)	50(102)	62(327)	54(453)	54(315)	50(535)
Chemical	White oak	67(46)	44(120)	64(213)	57(223)	60(222)	54(295)

Northern red oak seedling mortality was highest for small diameter trees. For instance 1,512 of the 2,491 seedlings tagged and measured were 0.1 inch in diameter, and 62 percent of these died. However, for seedlings greater than or equal to 0.2 inches in diameter, only 22 percent had died by the time of remeasurement. Of the 0.1 inch diameter seedlings, 97 percent were less than 1 foot tall, the height class most likely to die within 1 or 2 years. It is likely that many of these trees died prior to opening creation.

Due to the low sample size for post oak, hickory and black walnut, it was not possible to develop a survival model for them. Survival of white oak, black oak, black cherry, and ash showed no relationship with treatments using logistic regression. At this early stage, many of the regenerating trees consist of very small (less than 1 foot tall) seedlings that typically have high mortality rates and sporadic cycles of first-year seedlings being recruited into the regeneration pool.

Seedlings greater than 1 foot tall have successfully been used to examine the impact of silvicultural treatments on tree species dynamics (Loftis 1990). It is also possible that a response has not been observed due to the small size of our openings. Oak reproduction in small openings of less than 0.5 acres will have a relatively large proportion of the reproduction affected by the surrounding stand and could favor development of more shade tolerant species (Sander and Clark 1971, Smith 1981, Sander and others 1983). Thus, the relative capacity of oaks to compete may be diminished.

However, increasing public concern over visual impact of large openings on public lands will likely place further constraints on maximum opening size. To effectively manage uneven-aged upland hardwood stands and maintain an acceptable stocking of oaks and other desirable species, public land managers will require information on the minimum opening size for successful growth and development of reproduction. This information will provide the tools to help balance both the wants of their publics and the growth and development needs of the forest.

CONCLUSIONS

It was not possible to detect any change in survival of white oak, black oak, black cherry, and ash in relation to treatments over the 4-year period after overstory removal. At this time it is difficult to determine if the lack of response is due to the small size range of openings, natural high mortality of small seedlings, or the short time interval between overstory removal and the following regeneration measurements. Remeasurement of these trees during and after the stem exclusion phase of development should determine effectiveness of the treatments.

Future hardwood opening size studies should consider including somewhat larger openings. This would increase the probability of capturing the minimum opening size to successfully regenerate oaks and other species intermediate to intolerant of shade. Additionally, a remeasurement of regeneration close to the time of opening creation will help determine how much mortality has occurred prior to treatment, making analysis of short term post-harvest data easier to interpret.

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EFFECTS OF GAP SIZE, DURATION OF DAYLIGHT, AND PRESENCE OF LEAF LITTER ON FOREST REGENERATION

G. Andrew Bartholomay and Todd W. Bowersox[1]

ABSTRACT.—Selection systems are used to manage multi-cohort forest stands by removing individual and/or groups of trees to create 0.01- to 1.0-ha openings in the canopy. Inherent in the selection system are the dual roles of tending the residual trees and regenerating a new cohort of tree seedlings. Research of silvicultural selection systems has historically focused on the ability to maintain an appropriate residual stand structure (number of trees per cohort) in perpetuity. However, few studies have specifically addressed the regeneration phase raising questions about controlling species composition and development within the new cohort. The goal of this research was to determine if

1) the size of a forest gap,
2) the duration of daylight on areas within the gap, and
3) the presence or absence of leaf litter on the forest floor affected tree species recruitment and growth.

Two pairs of four different sized openings (0.02 ha, 0.04 ha, 0.10 ha, 0.20 ha) were created in a 4.5 ha, mixed hardwood, multi-cohort stand located at Penn State's Stone Valley Experimental Forest in central Pennsylvania. Recruitment inventory plots were located in each gap so that three areas of duration of daylight within each gap (0 to 1 hour, 2 to 3 hours, 4 to 5 hours) were sampled. The inventory plots were further divided into four subplots, two with leaf litter present and two without leaf litter. Results from the first growing season following harvest indicated that forest gap size and the duration of daylight within the gap significantly affected the stem density, average height, and average leaf spread of recruited shade intolerant tree species. Previously, only gap size had been considered important to the regeneration of multi-cohort stands. The recognition of duration of daylight effects implies that tree species stratification within a gap may occur along light exposure gradients not previously recognized. Future research will attempt to better explain the effect of daylight on regeneration through

1) continued measurements within the treated stand,
2) repeated experimentation in an additional stand, and
3) controlled light exposure experiments.

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CANOPY GAP REPLACEMENT FAILURE IN A PENNSYLVANIA FOREST PRESERVE SUBJECT TO EXTREME DEER HERBIVORY

Brian S. Pedersen and Angela M. Wallis[1]

While research has demonstrated the adverse effects of deer herbivory on forest regeneration in forests managed for timber production, less study has been devoted to the long term effects of deer on the dynamics of forests set aside as natural areas. At sufficiently high population densities, deer could interrupt the typical cycle of canopy gap formation and replacement by consuming all young trees. But other factors, such as invasive species, may also inhibit tree establishment and gap replacement.

We are comparing the canopy gap replacement process in two forest stands in south-central Pennsylvania (Perry County). At the private Florence Jones Reineman Wildlife Sanctuary (RWS, 1,375 ha), hunting has been prohibited since this preserve's establishment in the 1960s. A sharp browse line and frequent sightings of white-tailed deer (*Odocoileus virginianus*) suggest a high deer density. Asian stilt grass (*Microstegium vimineum* [Trin.] Camus) forms dense stands that exclude other plants. Native vines, principally grape (*Vitis* spp. L.), blanket the canopies of many understory trees. Our comparison stand is 19 km away in State Game Lands 170 (SGL, 3,550 ha), managed by the Pennsylvania Game Commission for wildlife and hunting. Both stands have a northerly aspect and elevations of 230-270 m.

At fixed intervals along line transects, we applied a plotless sampling method to determine the density and basal area of woody stems >1.30 m tall (number of trees sampled: 230 at RWS and 185 at SGL). We searched for woody stems 0.30 to 1.29 m tall in strip transects (strip transect areas: 950 m^2 at RWS and 300 m^2 at SGL). These trees represent the *null gap*, the overall forest composition, for each stand.

To quantify the influence of deer on future canopy trees, we examined null gap trees <30 cm dbh (diameter at breast height, 1.30 m) for buck rubs (cambial damage likely to be from abrasion by deer antlers; number of trees: 133 at RWS and 89 at SGL). We also examined these understory trees for the presence of vines in their canopies (number of trees: 133 at RWS and 90 at SGL).

The extent of canopy gaps was measured as the proportion of line transects intersecting gaps (line transect lengths: 1,820 m at RWS and 600 m at SGL). Asian stilt grass cover was measured as the proportion of line transects intersecting stands of this grass (line transect lengths: 950 m at RWS and 600 m at SGL).

To characterize the past canopy gap replacement process, we identified dead trees >20 cm dbh, large enough that they could have been part of the overstory, in strip transects and noted whether the forest canopy above these dead trees was closed (number of dead trees: 66 at RWS and 65 at SGL; strip transect areas: 7,600 m^2 at RWS and 4,800 m^2 at SGL). To estimate relative gap ages, we rated the state of decomposition of the dead trees on a 0 to 5 scale.

At RWS, we studied the vegetation within 14 gaps in detail. Along line transects we quantified Asian stilt grass cover (as above; line transect length: 580 m). We searched for woody stems 0.30-1.29 m tall in strip transects (strip transect area: 566 m^2). All understory trees >5 cm dbh within these gaps were examined for buck rubs (number of trees: 201) and vines (number of trees: 194).

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The density of null gap stems <20 cm dbh was nearly 20 times less at RWS (281 stems/ha) than at SGL (5,490 stems/ha), consistent with our hypothesis of extreme deer herbivory at RWS. For stems 0.30 to 1.29 cm tall alone, the difference in null gap stem density was over 50 fold: 84.2 stems/ha at RWS and 4,770 stems/ha at SGL. Further, all of the 0.30 to 1.29 cm tall stems sampled at RWS were witch hazel (*Hamamelis virginiana* L.), a species not capable of joining the forest overstory.

At least 90 percent of the 0.30-1.29 cm tall stems sampled at SGL were species capable of joining the forest overstory, including blackgum (*Nyssa sylvatica* Marsh.), 65 percent; oak (*Quercus* spp. L.), 10 percent; and sassafras (*Sassafras albidum* [Nutt.] Nees), 10 percent. In typical forests, the density of stems is substantially greater in canopy gaps than for the forest as a whole. However, at RWS the density of trees 0.30 to 1.29 m tall in the gaps (17.7 stems/ha) was less than for the null gap (84.2 stems/ha).

The two forest stands have comparable overstories; the density of trees >20 cm dbh was 154 stems/ha at RWS and 130 stems/ha at SGL. Total basal area was 17.6 m^2/ha at RWS and 19.3 m^2/ha at SGL. At RWS, five genera accounted for 83 percent of the basal area: tulip magnolia (*Liriodendron tulipifera* L.), 23 percent; maple (predominantly *Acer rubrum* L.), 20 percent; birch (*Betula lenta* L.), 15 percent; oak (including *Quercus alba* L., *Q. prinus* L., *Q. rubra* L., and *Q. velutina* Lam.), 13 percent; and hickory (including *Carya glabra* [Mill.] Sweet var. *glabra*, *Carya glabra* var. *odorata* [Marsh.] Little, and *Carya ovata* [Mill.] K. Koch, 13 percent). At SGL, three genera accounted for 83 percent of the basal area: oak, 54 percent; maple, 17 percent; and birch, 12 percent. The largest tree at RWS was 65.9 cm dbh and largest tree at SGL was 69.4 cm dbh.

We found evidence that past gap replacement has been less successful at RWS than at SGL. Despite the greater age of the gaps at RWS (average state of decomposition ratings of the dead trees: 3.6 at RWS and 2.7 at SGL), dead trees were more likely to still be in gaps at RWS (42 percent) than at SGL (31 percent). The RWS stand also had more gap area (12 percent) than the SGL stand (8 percent) even though the RWS stand had a lower density of dead trees (86.8 stems/ha) than SGL (135 stems/ha).

Our expectation of a higher deer density at RWS is supported by the greater frequency of buck rubs at RWS (15 percent) compared to SGL (8 percent). The trees within gaps at RWS were even more likely to be affected (23 percent) suggesting deer were preferentially using the gap habitat. To quantify relative deer density in our two stands, we will be measuring fecal pellet production during winter 2002 (and reporting the results at the conference). Other recent studies suggest the significance of deer herbivory at RWS. A previous deer fecal pellet production study found 1.6 deer/ha, over 20 times that recommended by the Pennsylvania Game Commission to insure forest regeneration. An 8 year exclosure study shows that trees capable of reaching the forest canopy can regenerate at RWS when protected from deer.

Asian stilt grass cover was greater at RWS (55 percent) than at SGL (0.0 percent; but Asian stilt grass was observed within exclosures at SGL). Within the 14 gaps at RWS, Asian stilt grass cover was slightly higher (62 percent). Similarly, vines influenced more null gap trees at RWS (35 percent) than at SGL (24 percent). Within the gaps at RWS, vines influenced 51 percent of the trees.

In conclusion, the canopy gap replacement process at RWS has been inhibited. While Asian stilt grass and vines are a major presence at RWS, these species are not solely responsible. Large areas of the forest floor are free of Asian stilt grass, even in gaps, and half of the understory trees within gaps are unaffected by vines. Rather, our findings, along with those of others, indicate that deer herbivory is responsible for the gap replacement failure. Without reductions in deer herbivory the future forest at RWS will become increasingly open as overstory trees die and are not replaced. However, reductions in deer herbivory at RWS may not lead to successful canopy gap replacement if Asian stilt grass spreads and prevents tree establishment even in the absence of deer browsing.

PATTERNS OF TREE SPECIES DIVERSITY AND COMPOSITION IN OLD-FIELD SUCCESSIONAL FORESTS IN CENTRAL ILLINOIS

Scott M. Bretthauer, George Z. Gertner, Gary L. Rolfe, and Jeffrey O. Dawson[1]

ABSTRACT.—Tree species diversity increases and dominance decreases with proximity to forest border in two 60-year-old successional forest stands developed on abandoned agricultural land in Piatt County, Illinois. A regression equation allowed us to quantify an increase in diversity with closeness to forest border for one of the forest stands. Shingle oak is the most dominant species in both stands. While other oak species importance values are lower in the stand with less forested border, shingle oak's importance value is higher, suggesting it has a seed dispersal mechanism that is different than the other oaks. Other dominant trees in the 60-year-old stands include American elm, slippery elm, and black walnut.

In agricultural regions of the Midwest there are few examples of undisturbed old-field succession to forest. Continuing disturbances such as mowing, grazing, and selective tree cutting during forest succession are common, and they disrupt natural successional patterns. At Allerton Park in Piatt County, Illinois, old agricultural fields set aside in the 1930s have reverted to forest without such disturbances. The configuration of these fields allows not only an opportunity to describe the woody species composition and forest structure of the approximately 60-year-old successional forests, but also to examine the effect of surrounding forest and old-field configuration on the composition and diversity of the successional forests.

Surrounding vegetation influences tree invasion during old-field succession (Bazzaz 1968, Myster 1993), and species found invading old-fields are often found in adjacent forest stands (Ashby and Weaver 1970, Buell and others 1971, Crowder and Harmsen 1998). The ability of a tree species to disperse seed onto a successional area is thought to be as important as other factors, such as competition, in determining species composition on old-fields (De Steven 1991a, 1991b). Density of trees on fields is greater near forest borders during the earlier periods of succession (Myster and Pickett 1992), and this relationship is more pronounced on smaller fields, which also exhibit greater species richness when in close proximity to forest border (Crowder and Harmsen 1998).

Differing seed dispersal mechanisms (wind, bird, or mammal) of trees are also important in determining the ability of trees to invade an old-field. These mechanisms affect the timing of invasion for individual species (Buell and others 1971, McDonnell and Stiles 1983, Christensen and Peet 1984, Burton 1989, Myster and Pickett 1992), and thus influence diversity by determining which species can become successfully established.

If forest border influences the species composition and densities of successional forests on old-fields, then different amounts of forest border along the perimeter of an old-field and the extent to which the forest surrounds the old-field should have an effect on tree species composition and diversity of successional forests. At Allerton Park, we studied two forest stands that developed on old-fields abandoned approximately 60 years ago with differing original amounts of forest border.

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We hypothesized that a forest stand that developed on a narrow field almost completely surrounded by forest would have greater species diversity than a forest stand that developed on a wider field that had forest border only along certain portions. We further hypothesized species diversity would decrease and dominance of the most common species would increase with increasing distance from forest border. We test these hypotheses and describe the species composition and structure of these successional forest stands.

STUDY SITE

Research was conducted at Robert Allerton Park in Piatt County, Illinois (Latitude: N39° 59.9′; Longitude: W88° 38.7′). The park is located along the Sangamon River and was donated to the University of Illinois at Urbana-Champaign in 1946 by Robert Allerton, who had maintained a mixture of forest, agricultural fields, and gardens on his estate. The park is located in the Prairie Peninsula of the oak-hickory forest region (Braun 1950).

Two separate upland forest tracts that were formerly devoted to agriculture and had differing percentages and configurations of forested border were selected. The first stand is located in the southwestern portion of the park and is bisected by a narrow grass corridor. Willow Branch, a tributary of the Sangamon River, runs through the southern portion of this stand, but the associated floodplain forest was not included in this study. The rest of the stand, the portion sampled, is at least 4.5 m above the floodplain of both the Sangamon River and Willow Branch and not subject to periodic flooding. The topography is fairly level with 0 to 7 percent slope. The soils in this stand consist of Miami loams and Xenia and Russell silt loams. They range from moderately well drained (Xenia) to well drained (Miami and Russell). These soils were formed under forest vegetation on till plains in loess, other silty material, and the underlying calcareous loam glacial till (Martin 1991). This stand is approximately 15.4 hectares in size and is referred to as stand 1.

The second stand is located in the eastern part of the park directly south of the Sangamon River, and is approximately 1,200 m northeast of stand 1. Slope ranges from 0 to 7 percent. The soils are in the same series as stand 1 with the addition of minor areas (less than 20 percent of total stand area) of Sunbury silt loam, which is somewhat poorly drained and was formed in loess and the underlying calcareous loam glacial till under forest vegetation (Martin 1991). The stand is at east 6 m above the floodplain of the Sangamon River, and the only areas likely to have standing water are those with the Sunbury silt loam. This second stand is approximately 16.2 hectares in size and is referred to as stand 2.

The critical difference between the stands for the purposes of this study is the amount of original surrounding forest and configuration of the fields. Stand 1 was almost completely surrounded by mature forest and is long and narrow (fig. 1). Stand 2 was bordered by mature forest only along its northern boundary and by sparse tree cover along the eastern side. Its remaining border was adjacent to additional abandoned pasture, and it is wider than stand 1. A previous study determined variability of upland soil properties at Allerton Park, including percent moisture and bulk density, to be low (David and Wang 1989).

While certain portions of Allerton Park were selectively cut for timber between 1847 and 1865 (Johnson and Bell 1975), both study stands have a history of past agricultural use. Local farmers cleared them of their original timber in the late 1800s, and records suggest both stands were used for cattle grazing (Foster 1981). The upper horizons of the soils were examined for signs of past agricultural disturbance. Both stands have signs of past grazing, and in portions there are indications of plowing. Indicators of past agricultural use include an abrupt soil structure change (granular to subangular blockly) at depths of 13 cm to 25 cm, nonpedogenic mixing of the A and B horizons, and evidence of compaction in the lower A horizon.

Aerial photographs from 1936 and maps from 1948 show that both stands were open grassland with a few scattered trees. Dykstra (1999) describes stand 1 as being a secondary forest and stand 2 as being an abandoned field in 1940. Both stands were described as successional forest in 1976 (Jones and Bell 1976). Since the University of Illinois assumed management of the Park in 1946, these stands have been allowed to succeed naturally without direct anthropogenic disturbance. Based on the aerial photographs and observations, it appears both stands began succession to forest in the mid to late 1930s.

Figure 1.—*Aerial photograph showing stands 1 and 2 in Allerton Park during the 1970s. The Sangamon River can be seen in the upper (north) portion of the photograph (Jones and Bell 1976).*

METHODS

The forest overstory in both stands was sampled using 0.081 ha circular plots systematically located 63.3 m apart at cardinal azimuths, resulting in a 20 percent inventory. There were 38 plots in stand 1 and 40 plots in stand 2. Within each plot, all woody stems greater than 6 cm diameter at breast height (dbh) were recorded. Dbh was measured to the nearest centimeter, and each stem was recorded by species in 1-cm diameter classes. Fieldwork was conducted in late summer and fall of 1998 for stand 1, and late summer and fall of 1999 for stand 2. The number of stems per ha, basal area in m^2 per ha, and frequency were calculated for each species within each stand.

Importance values (IV) were calculated using a sum of relative density, relative basal area, and relative frequency, and were scaled to 100 by dividing by three. Importance values were also calculated for each individual plot using only relative density and relative basal area. Distance to nearest forested border (m) was

determined by denoting plot location on the 1936 aerial photographs, and measuring to the nearest forested border that had at least 50 percent canopy closure.

A Shannon index of diversity ($H'=-\Sigma p_i \ln p_i$ where p_i equals the proportion of importance for the ith species) and Simpson index of dominance ($D=\Sigma p_i^2$ where p_i equals the proportion of importance for the ith species) were calculated for each stand using the importance values from individual plots based on relative density and relative basal area. Both the Shannon diversity index (H') and the Simpson index of dominance (D) were used because of differences between the indices in sensitivity to species richness and dominance. The Shannon index is biased towards species richness, while the Simpson index is biased towards dominance (Magurran 1988). The stands appeared to be different with respect to both of these attributes.

A jack-knifing procedure (Zahl 1977) was used to calculate the indices for each stand. This resampling procedure calculates the index n times, where n is the number of plots. Each time the index is calculated, one plot is removed from the sample. A series of jack-knife estimates (VJ) are created for each plot, and represent the diversity of the stand with an individual plot removed. A series of n pseudovalues (VP) are also created, and are reliable for significance testing (Zahl 1977). The mean of the VP values is the best estimate of the index. Jack-knifing an index of diversity allowed us to examine the influence of individual plots on the overall diversity of a stand without losing this overall diversity. Calculating an individual index value for each plot and then an average would not accurately reflect the true diversity of a stand because diversity is sensitive to sample size (Magurran 1988).

In accordance with our hypothesis, we expected plots close to the forest border to have a positive influence, plots far from the border to have a negative influence, and intermediate plots to have little influence on diversity of a stand. The stands were tested for differences with t-tests using the VP values for both H' and D. The VJ values from the jack-knifing procedure using H' were examined for relationships with distance from forest border (m). In addition, a linear regression analysis was used to statistically quantify the relationship of VJ with distance from forest border for the wider stand 2.

RESULTS

The Appendix lists all tree species encountered in either stand by both the scientific and common name. The best estimate (mean of VP values) of H' is 2.99 for stand 1 and 2.44 for stand 2, indicating a greater diversity for stand 1. An estimated D value of 0.13 for stand 2 indicates a greater dominance by the common species than observed in stand 1, which had an estimated D value of 0.06. A higher dominance index indicates a lower diversity because fewer species are common. Both indices indicate stand 1 is more diverse. The two stands are significantly different for both indices according to the results of the t-tests (for H': $df=75$; p-value<0.0001; for D: $df=49$; p-value<0.0001).

The plots of VJ values (H') and distance (m) showed a trend for stand 2, but not stand 1. When plots from stand 2 that were close to the forest border were removed from diversity calculations during the jack-knifing procedure, diversity decreased, and when plots far from the border were removed diversity increased, indicating plots closer to forested border had a positive influence on diversity and plots far from the border had a negative influence on diversity. This can be interpreted to indicate that plots closer to the forest border are more diverse.

The results of the linear regression analysis, where VJ=2.41+0.00011*Distance (m), shows a significant relationship between the VJ values and distance ($df=39$; p-value<0.001). The R^2 value is 0.30. Considering the many factors that influence forest succession and community structure, and thus the variation in the VJ values, the significant regression relationship with an R^2 of 0.30 suggest that distance to forest border is an important factor contributing to variance of diversity among the plots.

Table 1 shows the ranking of importance values for stand 1. Shingle oak has the highest IV, followed by slippery elm and American elm. Other oak species include black oak, white oak, northern red oak, and bur oak. Black walnut and white ash also rank high in IV. Four hickories are present, including shagbark hickory, pignut hickory, bitternut hickory, and mockernut hickory. Although no individual hickory species has a high IV, their combined IV is greater than any other individual species except shingle oak and slippery elm.

The number of stems per hectare in five size classes for stand 1 is shown in table 2. The smallest size class (6 to 10 cm) is composed

Table 1.–*Relative density (RD, number of trees for species/total number of trees), relative basal area (RBA, basal area (m^2) for species/total basal area), relative frequency (frequency of species/total frequencies) (RF), and importance value (IV (RD+RBA+RF)/3) for stands 1 and 2*

	Stand 1					Stand 2			
Species	RD	RBA	RF	IV	Species	RD	RBA	RF	IV
Quercus imbricaria	11.4	15.3	6.5	11.0	*Quercus imbricaria*	22.2	27.8	10.0	20.0
Ulmus rubra	12.2	5.2	7.1	8.2	*Ulmus americana*	22.4	17.7	10.0	16.7
Ulmus americana	10.9	4.3	7.8	7.7	*Juglans nigra*	6.0	8.8	8.2	7.7
Quercus velutina	5.4	11.0	5.1	7.2	*Sassafras albidum*	10.7	5.9	6.0	7.5
Fraxinus americana	7.7	6.5	6.7	7.0	*Prunus serotina*	5.8	6.7	8.0	6.8
Juglans nigra	5.9	9.9	5.1	7.0	*Ulmus rubra*	7.7	4.6	7.5	6.6
Quercus alba	3.1	11.3	3.6	6.0	*Acer saccharinum*	4.5	5.6	4.7	4.9
Sassafras albidum	8.6	3.4	4.9	5.6	*Quercus velutina*	2.4	5.9	6.5	4.9
Gleditsia triacanthos	3.6	7.6	5.1	5.4	*Celtis occidentalis*	3.6	2.6	6.5	4.2
Celtis occidentalis	5.2	3.1	5.1	4.5	*Gleditsia triacanthos*	1.5	5.8	3.5	3.6
Prunus serotina	3.7	2.7	5.8	4.1	*Crataegus spp.*	2.9	0.7	6.5	3.4
Quercus rubra	2.7	3.3	4.9	3.6	*Quercus rubra*	1.7	3.1	2.0	2.3
Quercus macrocarpa	1.5	3.3	3.1	2.6	*Fraxinus americana*	1.5	0.7	3.5	1.9
Cercis canadensis	3.1	0.8	3.3	2.4	*Malus spp.*	1.8	0.4	3.2	1.8
Carya ovata	2.0	1.7	3.3	2.4	*Cercis canadensis*	1.5	0.5	3.0	1.7
Carya glabra	2.0	3.2	1.8	2.3	*Quercus macrocarpa*	0.7	0.9	1.5	1.0
Acer saccharum	1.5	0.5	3.1	1.7	*Acer negundo*	0.7	0.5	1.7	1.0
Tilia americana	1.4	0.9	2.7	1.7	*Carya cordiformis*	0.6	0.3	2.0	1.0
Juniperus virginiana	2.4	0.7	1.8	1.6	*Tilia americana*	0.4	0.4	1.5	0.8
Carya cordiformis	1.2	1.1	2.4	1.6	*Carya ovata*	0.5	0.4	0.7	0.6
Carya tomentosa	0.9	0.9	2.9	1.6	*Quercus alba*	0.3	0.6	0.7	0.6
Acer saccharinum	0.4	0.6	1.6	0.9	*Acer saccharum*	0.2	0.1	1.0	0.4
Viburnum prunifolium	0.4	0.0	1.6	0.7	*Carya tomentosa*	0.1	0.1	0.5	0.2
Platanus occidentalis	0.3	0.9	0.7	0.6	*Viburnum prunifolium*	0.1	0.0	0.5	0.2
Crataegus spp.	0.8	0.2	0.9	0.6	*Carya glabra*	0.1	0.0	0.2	0.1
Morus rubra	0.3	0.3	0.9	0.5	*Maclura pomifera*	0.0	0.0	0.2	0.1
Acer negundo	0.7	0.2	0.4	0.4	*Morus rubra*	0.0	0.0	0.2	0.1
Fraxinus quadran gulata	0.4	0.0	0.4	0.3	*Cornus florida*	0.0	0.0	0.2	0.1
Fraxinus pennsylvanica	0.1	0.2	0.4	0.2					
Robinia psuedoacacia	0.1	0.3	0.2	0.2					
Maclura pomifera	0.2	0.2	0.2	0.2					
Prunus spp.	0.0	0.0	0.2	0.1					
Cornus florida	0.0	0.0	0.2	0.1					
Total	100	100	100	100	Total	100	100	100	100

See Appendix 1 for nomenclature

mostly of slippery elm and American elm, which combined, represent about 30 percent of the stems. Sassafras, shingle oak, white ash, and hackberry also have high numbers of stems in this size class. In the next size class (11 to 20 cm) the elms, shingle oak, and sassafras comprise a major portion of the total stems, along with white ash and black walnut.

In the 21 to 30 cm dbh size class shingle oak is the most dominant tree. Honeylocust also represents a large portion of stems in this size class (table 2). Shingle oak is the most common tree in the 31 to 40 cm diameter class. The elms are rare in this size class, with black oak and northern red oak both having more stems than the elms. In the largest size class, 41 cm and greater, black oak and white oak outnumber shingle oak, the third most common species. The oaks combined make up almost 69 percent of the stems in this size class. Black walnut and honey locust are also common. The total basal area for stand 1 is 24.0 m^2 per hectare, with 782 stems per hectare.

In stand 2, shingle oak and American elm are the two species with the highest IVs (table 1). Black walnut is third, but has less than half the IV of American elm. Black oak, white oak, bur oak, and northern red oak are also present in

Table 2.—*Density (number of trees per hectare), basal area (BA, m² per hectare), and frequency (number of plots in which species occurred/total number of plots) by size classes (dbh in cm), ranked in order of importance value (table 1) for stand 1*

	Density						BA	Freq.
Species	6-10cm	11-20cm	21-30cm	31-40cm	41+cm	>6cm	>6cm	>6cm
Quercus imbricaria	27.6	27.6	14.6	12.7	6.5	89.1	3.67	76.3
Ulmus rubra	54.6	32.5	7.2	1.0	0.0	95.3	1.25	84.2
Ulmus americana	49.7	28.3	5.2	1.6	0.0	84.9	1.03	92.1
Quercus velutina	9.4	12.4	7.2	5.5	7.5	41.9	2.66	60.5
Fraxinus americana	22.8	23.1	8.5	4.9	1.0	60.1	1.57	78.9
Juglans nigra	8.5	20.5	7.8	3.9	5.2	45.8	2.38	60.5
Quercus alba	4.6	6.8	3.6	1.6	7.5	24.1	2.70	42.1
Sassafras albidum	35.8	26.3	4.9	0.0	0.0	67.0	0.82	57.9
Gleditsia triacanthos	1.3	7.8	10.7	5.2	2.9	28.0	1.84	60.5
Celtis occidentalis	22.8	10.4	5.9	1.6	0.3	41.0	0.75	60.5
Prunus serotina	11.4	8.5	7.8	0.7	0.3	28.6	0.66	68.4
Quercus rubra	7.2	6.5	4.2	2.0	1.3	21.1	0.79	57.9
Quercus macrocarpa	4.9	2.6	1.3	0.7	2.3	11.7	0.80	36.8
Cercis canadensis	15.9	8.1	0.0	0.0	0.0	24.1	0.19	39.5
Carya ovata	6.2	4.9	2.9	2.0	0.0	15.9	0.42	39.5
Carya glabra	0.7	6.8	4.2	3.3	0.3	15.3	0.76	21.1
Acer saccharum	7.8	3.6	0.7	0.0	0.0	12.0	0.12	36.8
Tilia americana	5.2	3.9	0.7	1.0	0.0	10.7	0.23	31.6
Juniperus virginiana	10.4	7.8	0.3	0.0	0.0	18.5	0.18	21.1
Carya cordiformis	3.9	2.3	2.6	0.7	0.0	9.4	0.26	28.9
Carya tomentosa	2.9	2.0	1.3	1.0	0.0	7.2	0.21	34.2
Acer saccharinum	1.0	1.3	0.7	0.0	0.3	3.3	0.16	18.4
Viburnum prunifolium	3.3	0.0	0.0	0.0	0.0	3.3	0.01	18.4
Platanus occidentalis	0.0	0.7	0.3	0.7	1.0	2.6	0.22	7.9
Crataegus spp.	3.3	2.9	0.0	0.0	0.0	6.2	0.05	10.5
Morus rubra	0.7	0.7	1.3	0.0	0.0	2.6	0.07	10.5
Acer negundo	2.9	2.3	0.0	0.0	0.0	5.2	0.05	5.3
Fraxinus quadrangulata	2.9	0.0	0.0	0.0	0.0	2.9	0.01	5.3
Fraxinus pennsylvanica	0.0	0.0	0.3	0.3	0.0	0.7	0.05	5.3
Robinia pseudoacacia	0.0	0.0	0.3	0.7	0.0	1.0	0.08	2.6
Maclura pomifera	0.0	1.0	0.7	0.0	0.0	1.6	0.05	2.6
Prunus spp.	0.0	0.3	0.0	0.0	0.0	0.3	0.00	2.6
Cornus florida	0.3	0.0	0.0	0.0	0.0	0.3	0.00	2.6
Total	327.7	261.7	105.0	50.7	36.4	781.6	24.0	1181.6

stand 2, but have lower IVs than in stand 1. The same four hickory species from stand 1 are present in stand 2, but also have lower IVs. Other common species in stand 2 include sassafras, black cherry, and slippery elm. The ranking of seventh in IV for silver maple, a floodplain species, on the upland stand 2 is noteworthy. Species present on stand 2 are for the most part identical to stand 1, but with differences in IVs.

In terms of stems per hectare, shingle oak and American elm dominate the two smallest size classes in stand 2 (table 3); sassafras also has a large number of stems. Black walnut has few stems in the 6 to 10 cm size class, but represents a significant portion of the 11 to 20 cm size class. Silver maple has a large proportion of its total number of stems per hectare in this size class as well. Shingle oak and American elm dominate the 21 to 30 cm size class, and along with black walnut and silver maple make up almost 65 percent of the stems.

In the 31 to 40 cm size class, shingle oak is the most common species, followed by American elm (table 3). Black walnut is also common, and honey locust has its highest density in this size class. The largest size class (41+ cm) is dominated by shingle oak, and American elm has a much lower percentage of the total stems per hectare in this size class than in the lower size

Table 3.—*Density (number of trees per hectare), basal area (BA, m² per hectare), and frequency (number of plots in which species occurred/total number of plots) by size classes (dbh in cm), ranked in order of importance value (table 1) for stand 2*

	Density						BA	Freq.
Species	6-10cm	11-20cm	21-30cm	31-40cm	41+cm	>6cm	>6cm	>6cm
Quercus imbricaria	65.2	68.3	22.2	9.9	11.4	177.0	5.86	100.0
Ulmus americana	79.1	66.7	24.7	5.9	2.2	178.5	3.73	100.0
Juglans nigra	7.4	19.8	14.5	5.3	1.2	48.2	1.85	82.5
Sassafras albidum	38.9	39.5	5.3	1.2	0.3	85.2	1.24	60.0
Prunus serotina	12.0	20.1	8.6	3.7	1.5	46.0	1.40	80.0
Ulmus rubra	31.8	22.9	5.6	0.9	0.3	61.5	0.96	75.0
Acer saccharinum	7.1	15.4	10.5	1.9	0.9	35.8	1.18	47.5
Quercus velutina	3.1	6.2	4.9	1.5	3.1	18.8	1.24	65.0
Celtis occidentalis	13.6	9.9	4.0	0.6	0.3	28.4	0.54	65.0
Gleditsia triacanthos	0.3	3.1	1.9	4.0	3.1	12.4	1.22	35.0
Crataegus spp.	18.2	5.3	0.0	0.0	0.0	23.5	0.15	65.0
Quercus rubra	3.1	4.6	3.4	1.9	0.9	13.9	0.65	20.0
Fraxinus americana	6.2	5.3	0.6	0.0	0.0	12.0	0.15	35.0
Malus spp.	12.7	1.5	0.0	0.0	0.0	14.2	0.08	32.5
Cercis canadensis	8.0	3.1	0.6	0.0	0.0	11.7	0.11	30.0
Quercus macrocarpa	1.2	1.9	1.5	0.6	0.0	5.3	0.19	15.0
Acer negundo	3.4	1.9	0.0	0.6	0.0	5.9	0.10	17.5
Carya cordiformis	3.1	1.2	0.6	0.0	0.0	4.9	0.06	20.0
Tilia americana	1.2	1.5	0.6	0.0	0.0	3.4	0.08	15.0
Carya ovata	2.2	1.2	0.6	0.3	0.0	4.3	0.09	7.5
Quercus alba	0.6	0.6	0.3	0.6	0.3	2.5	0.13	7.5
Acer saccharum	0.9	0.0	0.3	0.0	0.0	1.2	0.01	10.0
Carya tomentosa	0.3	0.6	0.0	0.0	0.0	0.9	0.02	5.0
Viburnum prunifolium	0.6	0.0	0.0	0.0	0.0	0.6	0.00	5.0
Carya glabra	0.0	0.6	0.0	0.0	0.0	0.6	0.01	2.5
Maclura pomifera	0.3	0.0	0.0	0.0	0.0	0.3	0.00	2.5
Morus rubra	0.3	0.0	0.0	0.0	0.0	0.3	0.00	2.5
Cornus florida	0.3	0.0	0.0	0.0	0.0	0.3	0.00	2.5
Total	321.2	301.1	110.9	38.9	25.6	797.8	21.1	1005.0

classes. The total basal area is less than stand 1 at 21.1 m² per hectare and at 798 stems per hectare there are more trees per hectare than in stand 1.

DISCUSSION

The two successional stands are different with respect to diversity and dominance. Stand 1 is more diverse and is less dominated by the most common species. It had a greater percentage of border shared with existing forest during succession and a narrower configuration than stand 2, and this apparently contributed to the increased diversity of the succeeding forest by providing a proximal seed source.

In stand 2, diversity is highest close to the forest border and decreases as distance from the forest border increases. Small fields have increased species richness closer to a forest border (Crowder and Harmsen 1998). The increase in diversity with proximity to border is evident only for stand 2. This relationship is not seen in stand 1, probably due to its narrow configuration and high percentage of forested border. No one plot was far enough from a surrounding forest border for a decrease in diversity to occur. The greatest distance from a plot to forest border for stand 1 is 99 m, as opposed to 204 m for stand 2.

Tree stem density is highest near forest borders early in succession, but as succession progresses through time, tree density at distances further from the forest border increases, eventually stocking the entire field with trees (Myster and Pickett 1992). Tree species which could disperse seeds on the portions of an old-field further from the forest border could establish themselves early and have a competitive advantage over species whose seed arrives later. Close to the forest border, a greater number of species

can successfully disperse seed, thus increasing diversity.

De Steven (1991a, 1991b) postulated that differential seed rain may be as important a factor as competition and herbivory in determining the species composition of successional forests. De Steven tested the effects of existing vegetation cover and herbivory on the germination (De Steven 1991a), growth, and survival (De Steven 1991b) of five hardwood species on upland old-fields of the piedmont. Using the results, a prediction was made as to which three species should be most commonly found naturally invading old-fields. Of the three predicted species, only one was found as frequently as expected.

The influence of nearby seed sources was put forward as a possible explanation of the actual frequency of species, which differed from what was predicted. In southern Illinois, approximately 300 km south of Allerton Park, tree species dominant in old-fields apparently are derived from trees common in adjacent forests, but competition may also be an important determinant of success or failure in the establishment of certain species (Ashby and Weaver 1970). Even if a species has the life history traits necessary to survive and compete in a successional environment, a lack of seed dispersal onto an old-field will obviously prevent its occupation of the site.

Few species in our study occur in only one of the stands, and all these have low IVs (table 1). The majority of species are present on both stands, but have different IVs. With the exception of shingle oak, the oaks have a greater total IV, density, and dominance (BA) in stand 1. The same is true for the hickories. Black walnut is relatively equal in IV and density between stands (tables 1, 2, and 3). Considering that these species, with the possible exception of bitternut hickory, produce seeds commonly distributed by mammals (Graney 1990; Johnson 1990; Rogers 1990; Sander 1990a, 1990b; Smalley 1990; Smith 1990a, 1990b; Williams 1990), more surrounding forest cover with seed and associated dispersal agents would seem to increase the chances of these species successfully dispersing into the old-field.

Wind dispersed species are often the first to invade old-fields (Harper and others 1970, Buell and others 1971, Smith 1975, Christensen and Peet 1984, Myster and Pickett 1992, Crowder and Harmsen 1998). The reduced forest border conditions and wider configuration of stand 2 would have provided an advantage for the wind-dispersed species over mammal-dispersed species. Silver maple has a higher IV in stand 2, where it was most dominant on, but not restricted to, the Sunbury silt loam, a somewhat poorly drained soil. Its lower IV in stand 1 is likely due to a lack of such poorly drained soils, which would favor such floodplain species.

The combined elms increase in IV in stand 2 (table 1), and are distributed throughout the stand irrespective of the Sunbury silt loam. American elm has a higher IV and slippery elm had a lower IV than in stand 1. The forest adjacent to stand 2 transitions, within 30 m of its border, to floodplain forests of the Sangamon River, where American elm is the more common elm species (Bell 1974, 1980), and this might account for predominance of American elm over slippery elm in stand 2.

The composition of old growth and other forested areas not converted to agricultural use at Allerton Park has been well documented (Boggess and Geis 1967; Bell 1974, 1980), and these forest stands represent the predominant forest composition for Allerton Park and the seed source for both stands. The composition of an old growth forest stand immediately west of stand 1 was dominated by oaks and hickories in 1963, with white and black oak being the two most important species. Slippery elm had the third highest IV, with American elm sixth. Both elms had high densities in smaller diameter classes that strongly contributed to their IVs. Shingle oak was present, but not very common (Boggess and Geis 1967).

On a portion of Allerton Park northeast of stand 2, and other nearby forests, Bell (1974, 1980) found American elm and hackberry most dominant in transition zones between floodplains and uplands that receive limited flooding. Shingle oak was present and also had its highest IV in these transition zones. The most dominant upland oak species was white oak, followed by black oak.

Bell (1980) also sampled Hart Woods, another forest along the Sangamon River, located about 35 km northeast of Allerton Park. An inventory of Hart Woods (Johnson and others 1978) found white, black, and red oak to be the three most dominant species with respect to BA. American elm and slippery elm both had high densities, and had shown a substantial increase in smaller stems since the previous inventory in 1965

(Root and others 1971, Johnson and others 1978). The increase in smaller elms was from regeneration following the mortality of mature elms from Dutch elm disease and phloem necrosis (Root and others 1971, Johnson and Bell 1975). Results from the latest inventory of Hart Woods in 1995 (Shaffer and Edgington 2003) show both elm species to continue to be common in the smaller diameter classes, but absent from the larger ones. Shingle oak is present but not a dominant species.

In successional forests in northern Illinois (approximately 180 km northeast of Allerton Park), an early stand that was dominated by hawthorns (*Crataegus* spp.) changed to one in which the hawthorns shared dominance with American elm, and then succeeded to a stand composed of sugar maple, northern red oak, American elm, black cherry, and hawthorns (Bell and del Moral 1977). Eighteen years later, hawthorns were greatly reduced in importance due to heavy mortality, and sugar maple was the most dominant species, followed by American elm. These two species are replacing the northern red oak and black cherry (Bell 1997).

A small woodlot developed on a clearing in Edgar County, Illinois, approximately 80 km southeast of our study site, was composed primarily of large numbers of small white ash, elm, maple, and black walnut (McClain and Ebinger 1968). Oaks comprised the majority of stocking on small sawtimber and pole-sized timber stands originating from abandoned pasture on the Springfield plain of south central Illinois (Jokela and Sawtelle 1985).

In the Shawnee Hills of southern Illinois, persimmon and sassafras dominated an early shrub community, with winged elm (*Ulmus alata*), eastern redcedar, and *Rhus* spp. (Sumac) also present in 3 to 15 year fields. At 25 years, persimmon and sassafras dominated in dense thickets, with winged elm, eastern redcedar, black gum (*Nyssa sylvatica*), and hawthorns also present. The same general composition was found in 40-year fields, but three hickory species (pignut, shagbark, and mockernut), sugarberry (*Celtis laevigata*), shingle oak, and tulip tree were also present in the tree layer.

The length of time necessary for oak invasion to begin may have been dependent on the time necessary for conditions to develop that favor squirrels, the primary dispersal agents for oaks, and the amount of nearby mature oak-hickory forest, which serves as a seed source as well as attractive habitat for dispersal animals (Bazzaz 1968). This general pattern is similar to that of this study. However, the higher dominance of shingle oak in stand 2 does not fit this pattern. If shingle oak relies on forest border for seed dispersion, then a lower IV would be expected in stand 2 because of its lower percentage of forest border and wide configuration.

The high water use efficiency of shingle oak seedlings under drought conditions (McCarthy and Dawson 1990) relative to other oak species is an attribute of early successional species (Bazzaz 1979), and may be a factor in its dominance, allowing it to assert itself early in the initial woody colonization period. The natural habitat of shingle oak is wooded floodplains, but it occurs more frequently as a pioneer species along dry disturbed roadsides and hedgerows (Wagner and Schoen 1976). At Allerton Park, shingle oak is a common species in undisturbed forest transition zones between floodplains and uplands (Bell 1974, 1980), but it is more dominant in the disturbed, upland successional forests of this study. The weight of shingle oak acorns may offer it an advantage over wind dispersed seeds in penetrating plant litter, which can prevent seeds from reaching the soil surface (Jokela and Sawtelle 1985, Fowler 1986, Hamrick and Lee 1987).

Dispersal mechanisms may be the major key to shingle oak's dominance on both stands, especially stand 2. The blue jay (*Cyanocitta cristata* L.), an important long distance disperser for oak acorns, is known to cache acorns along forest boundaries and in open early successional habitat, and facilitates germination by burying acorns in the ground and under litter (Darley-Hill and Johnson 1981; Johnson and Webb 1989; Johnson and others 1993, 1997). Blue jays prefer smaller acorns (Darley-Hill and Johnson 1981, Johnson and Webb 1989, Scarlett and Smith 1991, Johnson and others 1997), and are know to take shingle oak acorns (Hermes 1991). This could explain the greater abundance of shingle oak in both stands compared to the other oaks present on the study sites, which all have larger acorns (Harlow and others 1996). Harrison and Werner (1984) suggested the blue jay as a possible agent for oak invasion on the successional habitat of their study.

The high densities of shingle oak relative to other species in the upper size classes and its high percent frequency (tables 2 and 3) suggest that it was one of the initial invaders in both

stands. Oaks and other mammal-dispersed species do not invade old-fields until later in forest succession, preceded by wind and bird dispersed species (Bazzaz 1968, Buell and others 1971, McDonnell and Stiles 1983, Christensen and Peet 1984, Myster and Pickett 1992, Myster 1993, Crowder and Harmsen 1998). Thus, it is likely that shingle oak seeds were dispersed onto the old-fields at Allerton primarily by the blue jay, rather than by mammals. This would explain the higher IV of shingle oak in stand 2, where all other oaks had a lower IV.

Another possible explanation for high shingle oak IV is mature pasture trees which could have provided a seed source within the field. Field observations reveal that two of the seven large trees identifiable in the 1936 photograph of stand 2 could have been shingle oak, based on the presence of two large shingle oaks in a plot located where two pasture trees can be seen in the photograph. These two trees, however, are less than 40 m from the forest border. A few additional pasture trees can be seen in the photograph to the west and south of stand 2, and it is possible that these are shingle oaks, but field observations could not confirm this. Large black oaks were found near the center of stand 2 in plots located where other pasture trees can be seen in the 1936 photograph, but this species did not exhibit a higher IV for stand 2.

It is unlikely sprouting from pre-disturbance forest stumps occurred in these stands because indications and records of disturbance suggest both stands were plowed and cropped after initial clearing, necessitating removal of stumps from the previous forest. Shingle oak does sprout (Dolan 1994), and through repeated mowing, grazing, or burning, shingle oaks that could have become established early in the subsequent pasture phase may have developed advanced root systems, leading to seedling sprouts after the pasture was abandoned. However, it is not known how long the stands were pastured before they were left to forest succession, or how other oaks may have similarly persisted as seedling sprouts from root systems established after clearing. White oak and black oak both sprout vigorously (Rogers 1990, Sander 1990b). Thus, we cannot ascertain for certain the proportion of the trees in the successional stands derived from seedling sprouts of root systems developed during the pasture phase relative to trees developed from seedlings established after pasture abandonment.

In other parts of central Illinois, shingle oak accounted for 17 percent of the trees on a previously grazed, poletimber-sized post oak flat, and 14 percent of the trees on a recently abandoned pasture (Jokela and Sawtelle 1985). On the recently abandoned pasture there were scattered mature oak pasture trees, but none were shingle oak. It was estimated that acorns for shingle oak and other oak species found invading the old-field but not represented among the pasture trees had to have traveled up to 500 m to reach the site, further supporting the hypothesis that blue jay dispersal is important for shingle oak.

The high IV of American and slippery elm in both stands fits the general trend of these two species in other forests along the Sangamon River, being most dominant in the smaller size classes (tables 2 and 3). A sampling of the understory in both stands (Bretthauer, unpublished data) shows a high number of seedling and sapling elms. This, combined with the intermediate shade tolerance of American elm (Bey 1990), suggests American elm may not have been one of the initial invaders but a later successional tree. Crowder and Harmsen (1998), however, found American elm to be one of the first tree species to invade old-fields in Canada.

On abandoned agricultural fields in southern Iowa, elms and oaks dominated the seedling- and sapling-size class (Crow and others 1994). American elm also invades tallgrass prairies in Kansas (Briggs and Gibson 1992). At Allerton Park, American elm represents 9 percent of the stems 41 cm and greater in stand 2 (table 3), suggesting it was one of the initial trees present on this site. The recruitment of American elm into larger size classes in these stands is probably limited due to Dutch elm disease. Bell (1997) found mortality to be the highest in American elm when tree diameter reached 26 cm, and found the growth rates of American and slippery elm to be greater in northern Illinois than at Allerton Park.

Proximity to forest border is associated with increased tree species diversity, decreased dominance of the most common species, and differences in species composition of successional forest stands developed on abandoned agricultural lands. Surrounding vegetation seems to be a key variable for explaining the occurrence of tree species that invade old-fields, and should be considered as such together with competition and each individual species'

ecological amplitude when attempting to identify mechanisms underlying successional processes. Shingle oak is a dominant tree on both stands, and is more dominant in stand 2 than other oak species bearing larger seeds. A plausible explanation for shingle oaks ability to invade old fields more independently of near proximity to seed source is the documented preferential dispersal of its acorns by blue jays. Our results are indicative of the complexities and interactions inherent in old-field succession in Central Hardwood forests.

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APPENDIX 1: SCIENTIFIC AND COMMON NAMES OF TREE SPECIES

Scientific name	Common name
Acer negundo L.	box elder
Acer saccharinum L.	silver maple
Acer saccharum Marsh	sugar maple
Carya cordiformis (Wangenh.) K. Koch.	bitternut hickory
Carya glabra (Mill.) Sweet	pignut hickory
Carya ovata (Mill.) K. Koch	shagbark hickory
Carya tomentosa (Poir.) Nutt.	mockernut hickory
Celtis occidentalis L.	hackberry
Cercis canadensis L.	eastern redbud
Cornus florida L.	flowering dogwood
Crataegus spp. L.	hawthorn species
Fraxinus americana L.	white ash
Fraxinus pennsylvanica Marsh.	green ash
Fraxinus quadrangulata Michx.	blue ash
Gleditsia triacanthos L.	honeylocust
Juglans nigra L.	black walnut
Juniperus virginiana L.	eastern redcedar
Maclura pomifera (Raf.) C. K. Schneid.	osage-orange
Malus spp. Mill.	apple species
Morus rubra L.	red mulberry
Platanus occidentalis L.	sycamore
Prunus serotina Ehrh.	black cherry
Prunus spp. L.	cherry, plum species
Quercus alba L.	white oak
Quercus imbricaria Michx.	shingle oak
Quercus macrocarpa Michx.	bur oak
Quercus rubra L.	northern red oak
Quercus velutina Lam.	black oak
Robinia pseudoacacia L.	black locust
Sassafras albidum (Nutt.) Nees	sassafras
Tilia americana L.	American basswood
Ulmus americana L.	American elm
Ulmus rubra Mühl.	slippery elm
Viburnum prunifolium L.	black haw

THE POPULATION DYNAMICS OF GOLDENSEAL BY HABITAT TYPE ON THE HOOSIER NATIONAL FOREST

S.P. Meyer and G.R. Parker[1]

ABSTRACT.—Goldenseal (*Hydrastis canadensis* L.) is an herbaceous species found throughout the central hardwood forest ecosystem that is harvested from the wild for the medicinal herb trade. A total of 147 goldenseal populations were classified according to the Ecological Classification Guide developed for the Hoosier National Forest, and change in population size was calculated where data from 1998 was available. The majority of the populations were located on beech-sugar maple/Jack-in-the-Pulpit (*Fagus L.-Acer saccharum* Marsh./*Arisaema* Mart.) mesic slope, a common site in the Hoosier National Forest. The remeasured populations were an average of 40 percent larger, an unexpectedly large increase in a species noted for slow growth. Illegal harvesting of goldenseal may still be occurring on certain areas. Sustained population studies of this species are needed to better understand its biology and status in the wild.

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RESPONSE OF PIN CHERRY TO FIRE, CANOPY DISTURBANCE, AND DEER HERBIVORY ON THE WESTVACO WILDLIFE AND ECOSYSTEM RESEARCH FOREST

David W. McGill, Rachel J. Collins, and Walter P. Carson[1]

ABSTRACT.—We studied the impact of fire, canopy disturbance, and deer herbivory on the germination and development of pin cherry in four Appalachian hardwood stands located on the Westvaco Wildlife and Ecosystem Research Forest in Randolph County, West Virginia. Plots with simulated gaps and woven-wire fences were used to evaluate impacts of light and deer on regeneration. All four stands were burned in spring 1999. One month following prescribed fire, average initial pin cherry densities ranged from 23,500 seedlings ha^{-1} on plots that had gaps but no fences to 63,000 seedlings ha^{-1} on plots without either fences or gaps. By the end of the second growing season, seedling densities ranged from a low of 12,750 seedlings ha^{-1} in plots with gaps but no fences to 28,500 seedlings ha^{-1} on plots with gaps and fences. Maximum heights in fenced plots exceeded 2 m. We found that characteristic attributes of understory pin cherry communities in our study area are principally a function of deer herbivory.

Improved timber production in high-valued natural hardwood stands can be achieved by reducing the proportion of undesirable species that are present in stands, thereby assuring that all the stems in a stand are producing wood of high quality. Precommercial release and thinning operations are employed to eliminate or reduce these unwanted species from high quality stands (Church 1955). In some cases, stands can develop such heavy components of undesirable species that precommercial operations are forced to focus on eliminating the undesirables instead of selecting crop trees of exceptional grade. In northern and high-elevation regions of the Central Hardwood forest, pin cherry (*Prunus pensylvanica* L.f.) is an undesirable species for timber production, but it is one that can rapidly and aggressively acquire valuable canopy growing space in regenerating hardwood stands. Thus, control of the species is of interest to managers and landowners interested in maximizing timber revenues from forest properties.

Pin cherry has a complex life history. It is a pioneer species, highly intolerant of shade, and requires full sunlight for maximum growth (Wendel 1990). When open conditions are present, as they are in clearcuts, or following catastrophic wind damage, pin cherry grows faster than most species, even its congeneric black cherry (*Prunus serotina* Ehrh; Ristau and Horsley 1999). Hence, pin cherry has the ability to be the first species to position its crowns in the main canopy. Aiding in this rapid acquisition of canopy growing space is pin cherry's capacity to sucker from roots. We have observed suckers to grow 3 to 5 feet in the first season and occupy over 200 m^2 per clone following severance of the parent stem.

In addition to the capacity to sucker and dominate large areas in colonies, pin cherry begins flowering at very early ages. Seeds can persist in a dormant seed bank for 50 to 75 years (Marquis 1975, Wendel 1990) in various states of dormancy (Tierney and Fahey 1998). Following intense canopy disturbance or fires, tens of thousands of seeds per hectare germinate (Marquis 1975, Graber and Thompson 1978, Wendel 1987, McGill and others 1999). Further, Auchmoody (1979) found that nitrate

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nitrogen can stimulate the germination of these dormant seeds and proposed that repeated fertilizer applications might eventually deplete the persistent seed bank.

The purpose of this study is to examine the effects of multiple disturbances on fire cherry germination and development on the Westvaco Wildlife and Ecosystem Research Forest with the intent to learn more about factors that could lead to better pin cherry control strategies. In this paper our primary objective is to characterize patterns of pin cherry seed germination and seedling development under the effects of disturbance in the form of fire, canopy mortality, and deer herbivory.

METHODS

We monitored pin cherry germination and development over two growing seasons on plots established by Collins and Carson (1997) on the Westvaco Wildlife and Ecosystem Research Forest in Randolph County, West Virginia. The Westvaco Research Forest falls within the Allegheny Mountains Section of the Central Appalachian Broadleaf Forest (McNab and Avers 1994) and the vegetation therein is classified as mixed-mesophytic (Braun 1950). Dominant overstory species often include sugar maple (*Acer saccharum* Marsh.), basswood (*Tilia americana* L.), yellow-poplar (*Liriodendron tulipifera* L.), red maple (*A. rubrum* L.), and northern red oak (*Quercus rubra* L.). Red oak site indices 50 average about 70-75 on mesic sites.

Specifically, the experimental plots we used were a subset of a larger study directed at assessing factors affecting oak regeneration in central Appalachian hardwood stands (Collins and Carson 1997). This study has as a main factor "burning", hence, each of the four stands has a burned half and an unburned half. The subset that we used to monitor pin cherry comprised the half of each stand that was subjected to prescribed fire. Fires were applied to two stands on 1999 April 30 and two stands on 1999 May 4. Further discussion of this study applies only to the work and measurements in these burned stands.

Measurements

In 1998, prior to the application of prescribed fire, four experimental 20 m X 20 m (400 m^2) plots in each of two blocks per stand were laid out for a total of 32 plots. Four treatments were applied to these plots:

1) Fenced and simulated gap (F×G),
2) Fenced and no gap (F×NG),
3) No fence and simulated gap (NF×G), and
4) No fence and no gap (NF×NG).

The simulated gaps were created in the spring of 1998 by a combination of double girdling with a chainsaw and tree injection using herbicide (glyphosate) on trees with diameters at breast height (dbh) greater than 10 cm. Gaps were designed to be roughly 18 m in diameter, or 250 m^2. Injected trees were located in the center of each plot. Average basal area in these stands was about 29 m^2 ha^{-1} with gaps reduced to about 9 m^2 ha^{-1}. Deer proof fences were erected around 16 plots, two in each block, immediately following the girdling and herbicide treatments. Fences are 2 m tall and constructed with woven-wire fencing reinforced with rebar and metal posts.

In each of the thirty-two, 400 m^2 main treatment plots are five internal, and randomly placed 1 m^2 plots designed to subsample seedling regeneration. On these plots, we measured the heights and age of pin cherry seedlings that germinated as a result of the spring 1999 fires. In this paper, we report our observations from four dates in the 1999 growing season and three dates in the 2000 growing season.

Numerical Analyses

Graphical analysis and analysis of variance were used to characterize the patterns of and factors affecting pin cherry germination and development. We used a blocked, two factor factorial model with interaction to assess differences in pin cherry parameters of interest. Our two main factors (both binary) are 1) fence and 2) gap, and are randomized within two blocks nested in each of four stands. We carried out our statistical tests using SAS (SAS Institute, Cary, NC). F-tests resulting in p-values less than 0.10 were deemed statistically significant. Spreadsheets were used for graphical analyses.

Response parameters of interest included the number of seedlings ha^{-1}, mean seedling height, sum of seedling heights, and average maximum seedling height. These parameters are used in different applications to characterize populations of extremely diverse regenerating natural hardwood communities. For example, the sum of heights is a commonly used measure that combines the number of seedlings with height in order to quantify hardwood competition in plantation studies; we found it to be useful in this study as well. We compared these attributes of

young pin cherry populations at critical points over the two growing seasons that we monitored for this study. These critical points include the end of the first growing season (October 1999), the beginning of the second season (June 2000), and the end of the summer of the second growing season (August 2000).

RESULTS AND DISCUSSION

Pin cherry germination in the understories of our four research stands was evident only a few weeks following the prescribed fires in the spring of 1999. At the June 1999 measurement, average pin cherry germination densities were between 23,500 seedlings ha^{-1} (NF×G treatment) and 63,000 seedlings ha^{-1} (NF×NG treatment) with the overall average density equal to about 44,000 seedlings ha^{-1} (fig. 1). The maximum density observed in the NF×NG treatment, however, plummeted rapidly throughout the first growing season, ending the season with fewer seedlings (9,750 seedlings ha^{-1}) than all but the NF×G treatment.

Both fenced treatments maintained the most stable pin cherry seedling densities throughout the first growing season. The F×G treatment density averaged 55,250 seedlings ha^{-1} at the first measurement (June 1999) and decreased to 43,750 seedlings ha^{-1} by October 1999. Even considering the reduction, this treatment had the highest number of seedlings by season's end (fig. 1). The F×NG treatment finished the season ranking second highest in pin cherry seedling density despite the lack of a canopy gap to provide light and support vigorous growth of this highly intolerant species.

Seedling mortality during the first growing season is evident in the steep decline in the density levels for the unfenced treatments (fig. 1). Without fences to protect the young seedlings, most were browsed and we observed many that had been pulled up by the roots, likely dropped from herbivores' mouths as they browsed on the sparse, post-fire understory vegetation.

Pin cherry performance following germination has been exceptional on the F×G treatment plots. Figures 2 through 4 show the development of pin cherry seedlings using variables that characterize size development of this regeneration component; the F×G treatments clearly led the way in size development. Mean height of pin cherries (fig. 2) serves to show that F×G treatments had taller seedlings than other treatments through the 2 years, but the attribute does not well characterize the condition we

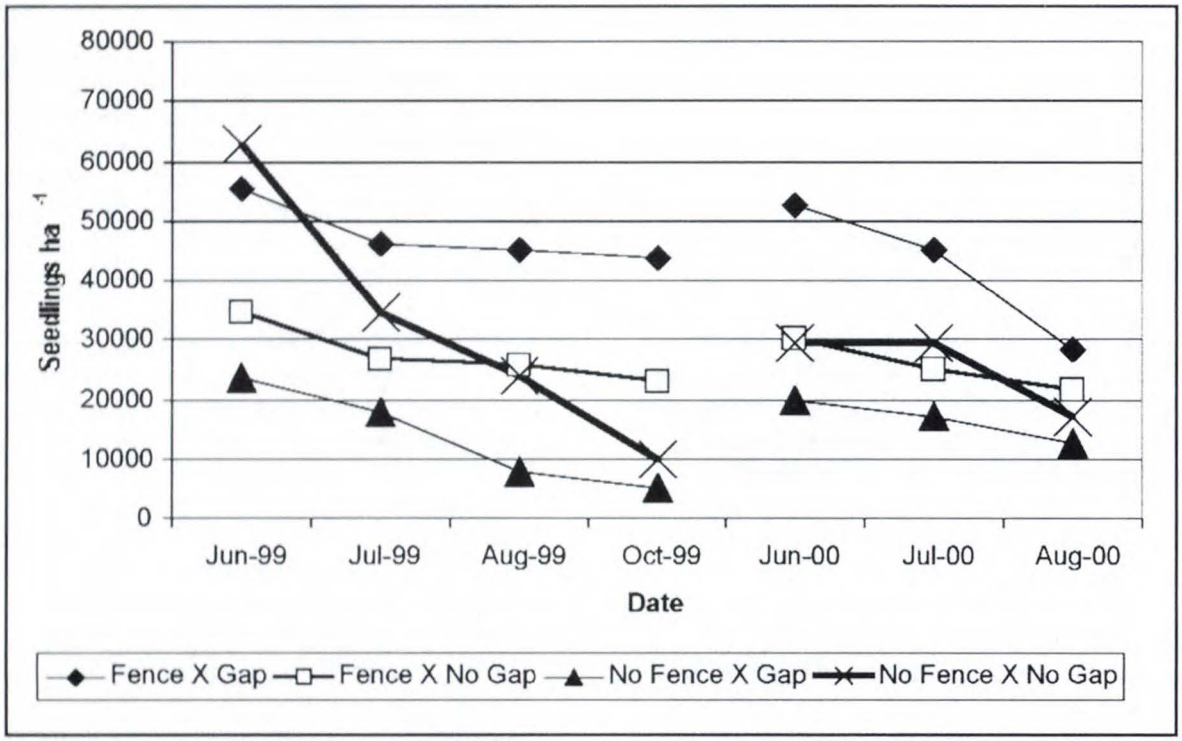

Figure 1.—*Understory pin cherry seedlings per hectare over two growing seasons on burned treatment plots. For each treatment there are eight observations, one per 400 m² treatment plot. Treatment plots are located in four stands on the Westvaco Research Forest in Randolph County, West Virginia.*

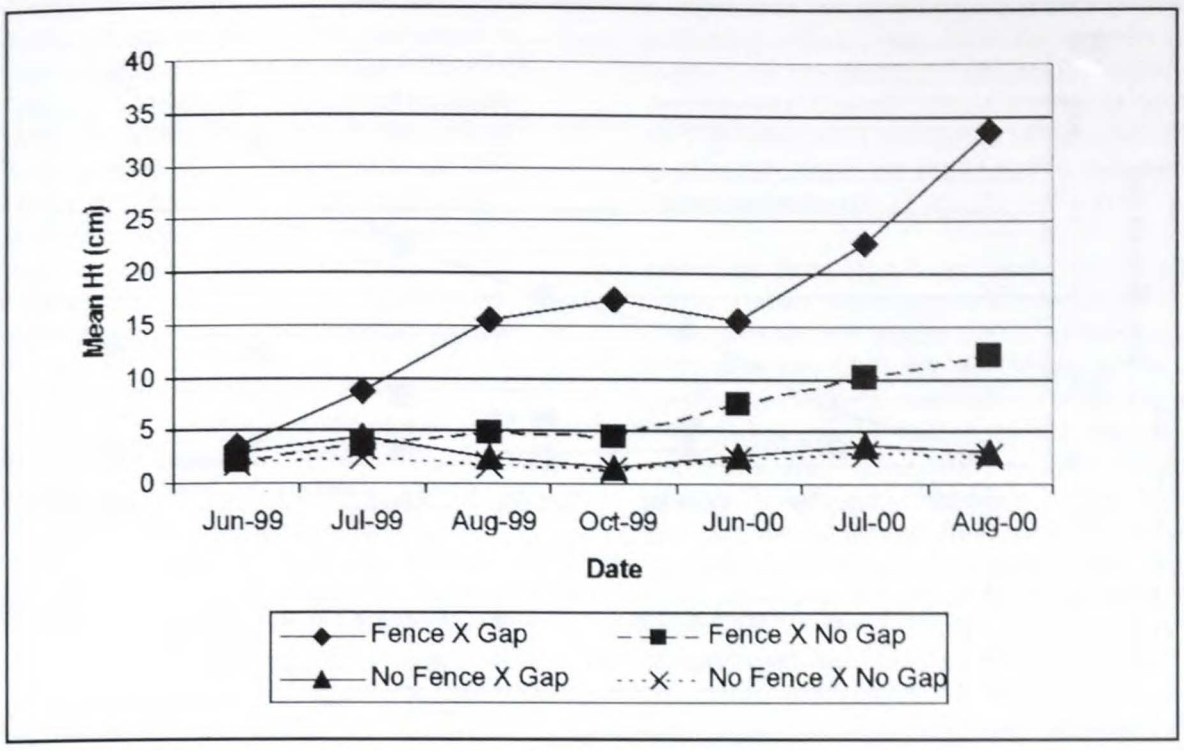

Figure 2.—*Mean height of understory pin cherry seedlings over two growing seasons on burned treatment plots. For each treatment, mean height is an average from eight 400 m² treatment plots.Treatment plots are located in four stands on the Westvaco Research Forest in Randolph County, West Virginia.*

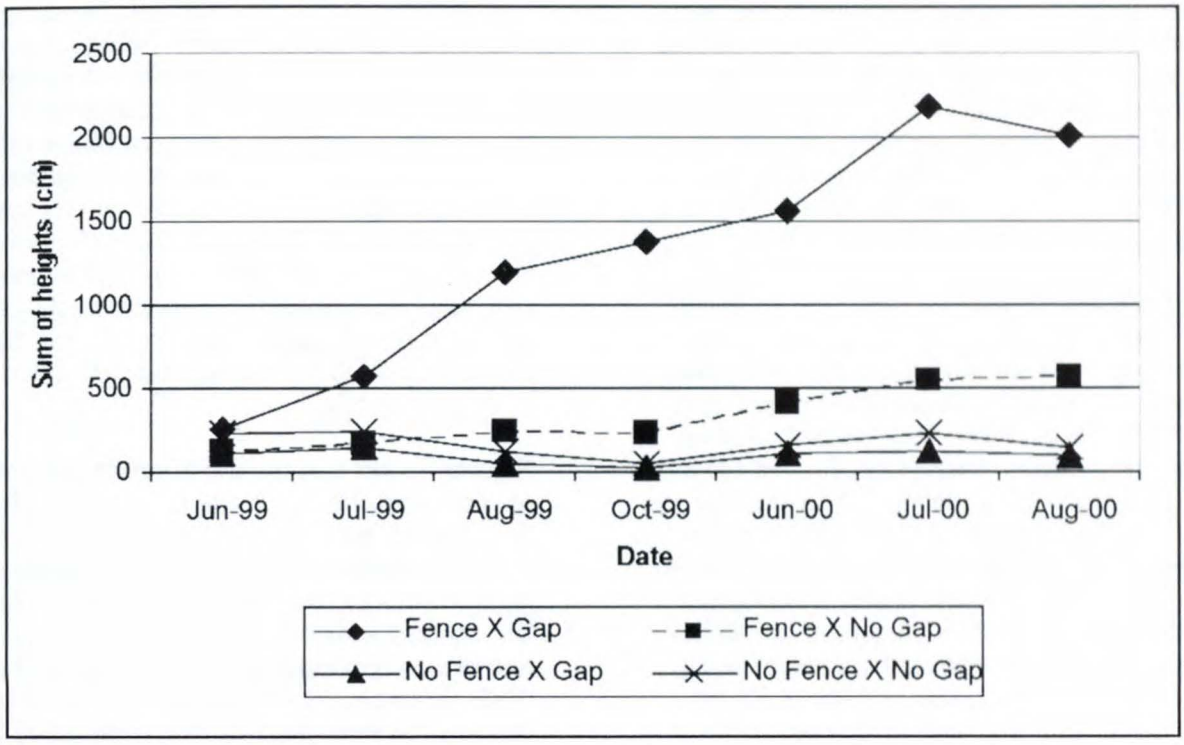

Figure 3.—*Mean sum of heights of understory pin cherry seedlings over two growing seasons on burned treatment plots. For each treatment there are eight observations, one per 400 m² treatment plot. Treatment plots are located in four stands on the Westvaco Research Forest in Randolph County, West Virginia.*

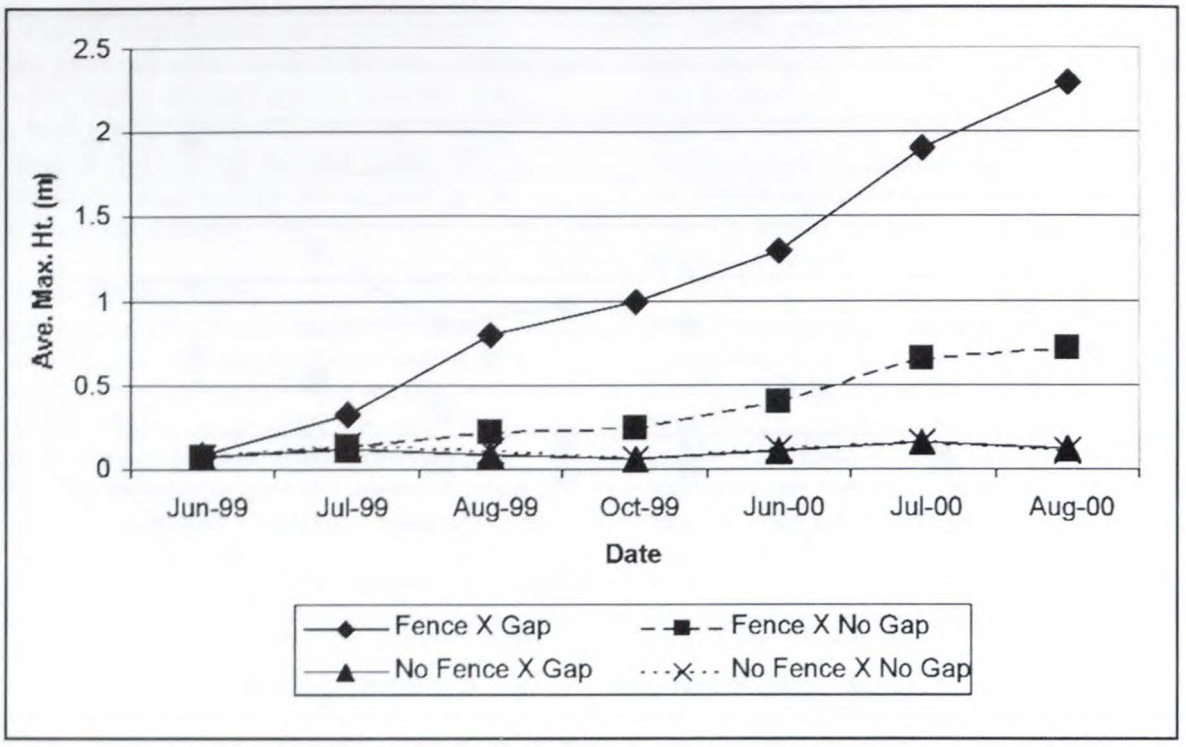

Figure 4.—*Mean maximum height of understory pin cherry seedlings over two growing seasons on burned treatment plots. For each treatment there are eight observations, one per 400 m² treatment plot. Treatment plots are located in four stands on the Westvaco Research Forest in Randolph County, West Virginia.*

found on these plots. The sum of the heights better exemplifies the abundance with the average summed heights exceeding 20 m at the end of the second growing season (fig. 3). At this time, the average maximum height of the pin cherry seedlings was well over 2 m in the F×G treatment plots (fig. 4). The F×NG treatment lagged at a distant second in this parameter, averaging 72 cm.

In spite of the seemingly clear pattern of development in our observed understory pin cherry densities and size attributes over the two growing seasons, we found an enormous amount of variation in pin cherry seedling density among the four study stands. Stands accounted for a significant amount of the variation in pin cherry seedling variables. Statistically, apart from the differences in pin cherry germination and development seen among stands, fencing explained significant variation in four variables for measurements made at the end of the first growing season (table 1). Fencing was a significant factor at the beginning and end of the second growing season only for mean height and average maximum height. In all cases where fencing resulted in significant hypothesis tests, fenced plots exceeded unfenced plots in pin cherry population density and size attributes.

At the beginning of the second growing season, we counted the number of new seedlings that emerged. Pin cherry exhibits epigeal germination with its cotyledons displayed above the soil surface. This feature is not only a distinguishing factor to separate pin from black cherry, but it is also evidence that a small seedling is a new germinant. Unfortunately, the presence of cotyledons is ephemeral, and they either wither and abscise rapidly, or are browsed. Consequently, distinction between 1- and 2-year-old seedlings in the second year became more uncertain in the second and third remeasurement occasions.

Figures 5 and 6 show the approximate proportions of new germinants in the second growing season. Again, without tags for individual seedlings it is impossible to say the exact number of new germinants in the latter two measurements of the second growing season. However, percentage of new germinants ranged from 35 percent (F×G treatment) to 67 percent (NF×G treatment) of the total seedlings at the June 2000 measurement. The F×G treatment did have the lowest percentage of new seedlings, but this represented an average of 18,500 new seedlings ha^{-1}.

Table 1.—Probability values (p-values) for testing factors affecting pin cherry germination and establishment in burned stands on the Westvaco Research Forest in Randolph County, WV. Response variables include number of seedlings ha^{-1}, mean height, sum of heights, and average maximum height for all pin cherry seedlings recorded at the three listed measurement dates. Measurement dates correspond to the end of the first growing season, the initial measurement of the second growing season, and the late summer measurement of the second growing season.

Source of Variation	Date: October 1999	June 2000	August 2000
	Seedlings ha^{-1}		
Stand	0.048	0.012	0.010
Block(Stand)	0.159	0.218	0.384
Fence	0.035	0.252	0.288
Gap	0.487	0.661	0.884
Fence X Gap	0.284	0.280	0.551
	Mean height (cm)		
Stand	0.011	0.003	0.009
Block(Stand)	0.846	0.767	0.856
Fence	0.014	0.024	0.026
Gap	0.077	0.188	0.239
Fence X Gap	0.090	0.218	0.260
	Sum of heights (cm)		
Stand	0.176	0.069	0.066
Block(Stand)	0.396	0.390	0.608
Fence	0.097	0.102	0.122
Gap	0.213	0.283	0.356
Fence X Gap	0.198	0.241	0.317
	Average max. height (m)		
Stand	0.016	0.006	0.011
Block(Stand)	0.686	0.643	0.822
Fence	0.021	0.030	0.026
Gap	0.110	0.202	0.250
Fence X Gap	0.131	0.250	0.270

The appearance of pin cherry in understories following fire, points to germination mechanisms similar to those found in certain California chaparral species by Keely and Frotheringham (1997). They suggest that oxides of nitrogen like those generated in wildfires initiate seed germination of a number of chaparral species; this is suggestive of a timing mechanism to key germination to correspond with an open canopy (eliminated through fire). The researchers go on to suggest that similar compounds emitted by automobiles and other polluting agents, may have an unseen negative impact on the semiarid chaparral ecosystems of the western United States. Untimely germination of succulent seedlings in otherwise dry stands of chaparral may lead to the demise of these individuals.

Likewise, fires that burn under intact canopies will stimulate dormant pin cherry seeds to germinate (McGill and others 1999). In the presence of high deer populations, germinating pin cherry seedlings will likely succumb to browsing and eventual mortality. Hence, under management scenarios where pin cherry is a problematic species, a light prescribed fire or fires that stimulate pin cherry germination beneath these intact canopies may bring about a depletion or at least a reduction in both the pin cherry seedbank and in future pin cherry densities. Prescriptions using similar methods should consider the potentially harmful impact of fire on high valued crop trees. Finally, attention to pulpwood products should also be considered since carbon can be detrimental in certain manufacturing processes like papermaking.

CONCLUSIONS

1) Deer herbivory was the key factor in reducing pin cherry densities and hampering development of seedlings over two growing seasons.
2) Germination of new pin cherry seedlings will occur in stands for at least 2 years following a spring prescribed fire. Management prescriptions targeting depletion of pin cherry in stands of high-valued species should heed this fact and wait at least 2 to 3 years following a fire before applying regeneration methods.
3) Pin cherry seedlings, protected by fences, continued to develop and were nearly beyond the range of deer browsing near the end of the second growing season. Hence, in stands where fires result in conditions similar to those conducted in this study and have low deer populations, understory pin cherry is likely to reach the stage of development where seed production begins (age 2 to 4).
4) Without protection from deer herbivory, pin cherry seedbanks that germinate under intact or mostly intact canopies will likely succumb due to both high shade and deer browsing. This conclusion suggests the possibility for prescriptions that call for low intensity prescribed fires to deplete pin cherry seedbanks prior to stand regeneration.

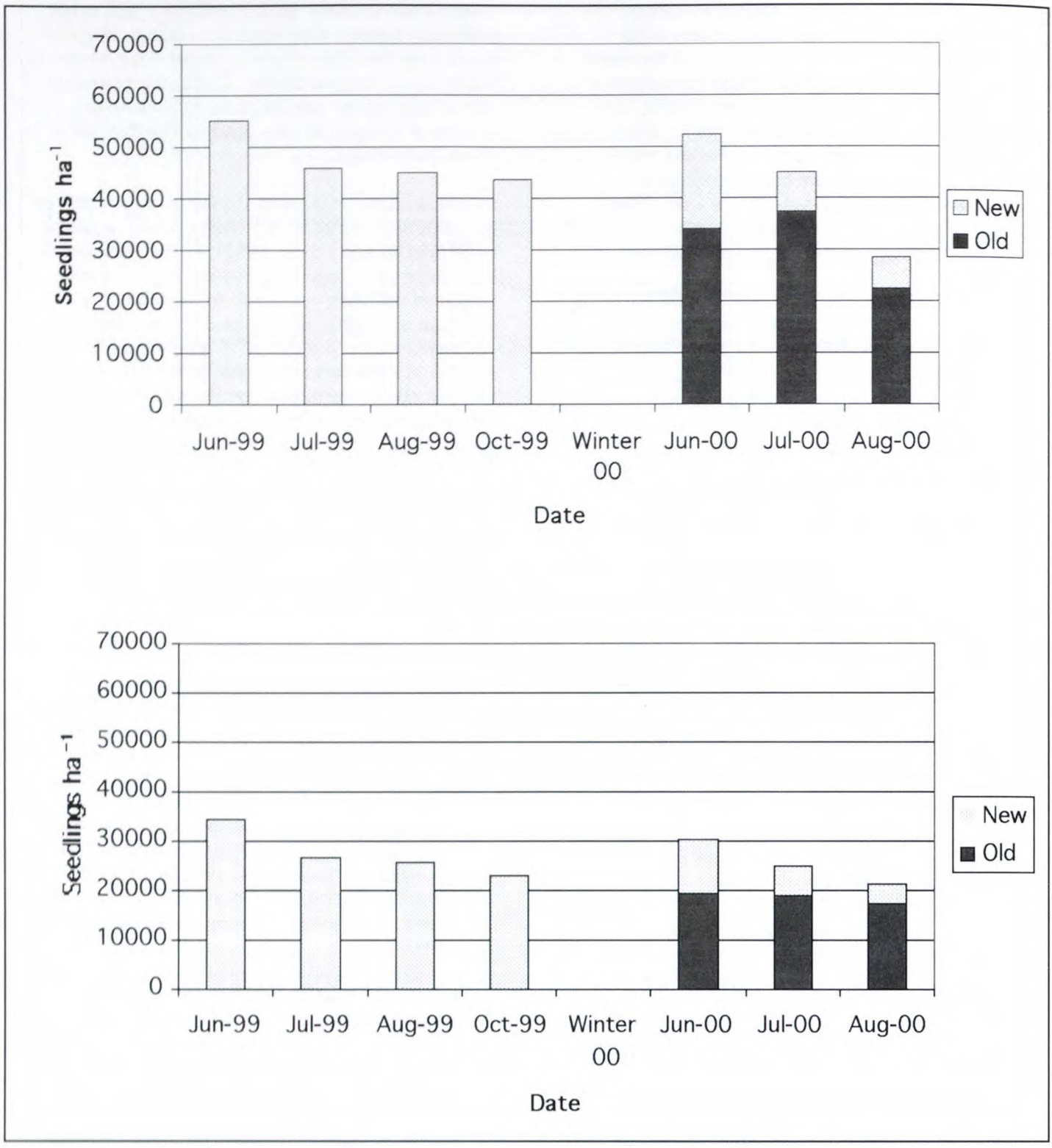

Figure 5.—*Number of pin cherry seedlings by age over two growing seasons on F×G and F×NG treatment plots. "New" seedlings are germinants from the growing season in which the measurement took place. "Old" seedlings germinated in during the first growing season. For each treatment, "seedlings per hectare" is an average of eight 400 m² treatment plots. Treatment plots are located in four stands on the Westvaco Research Forest in Randolph County, West Virginia.*

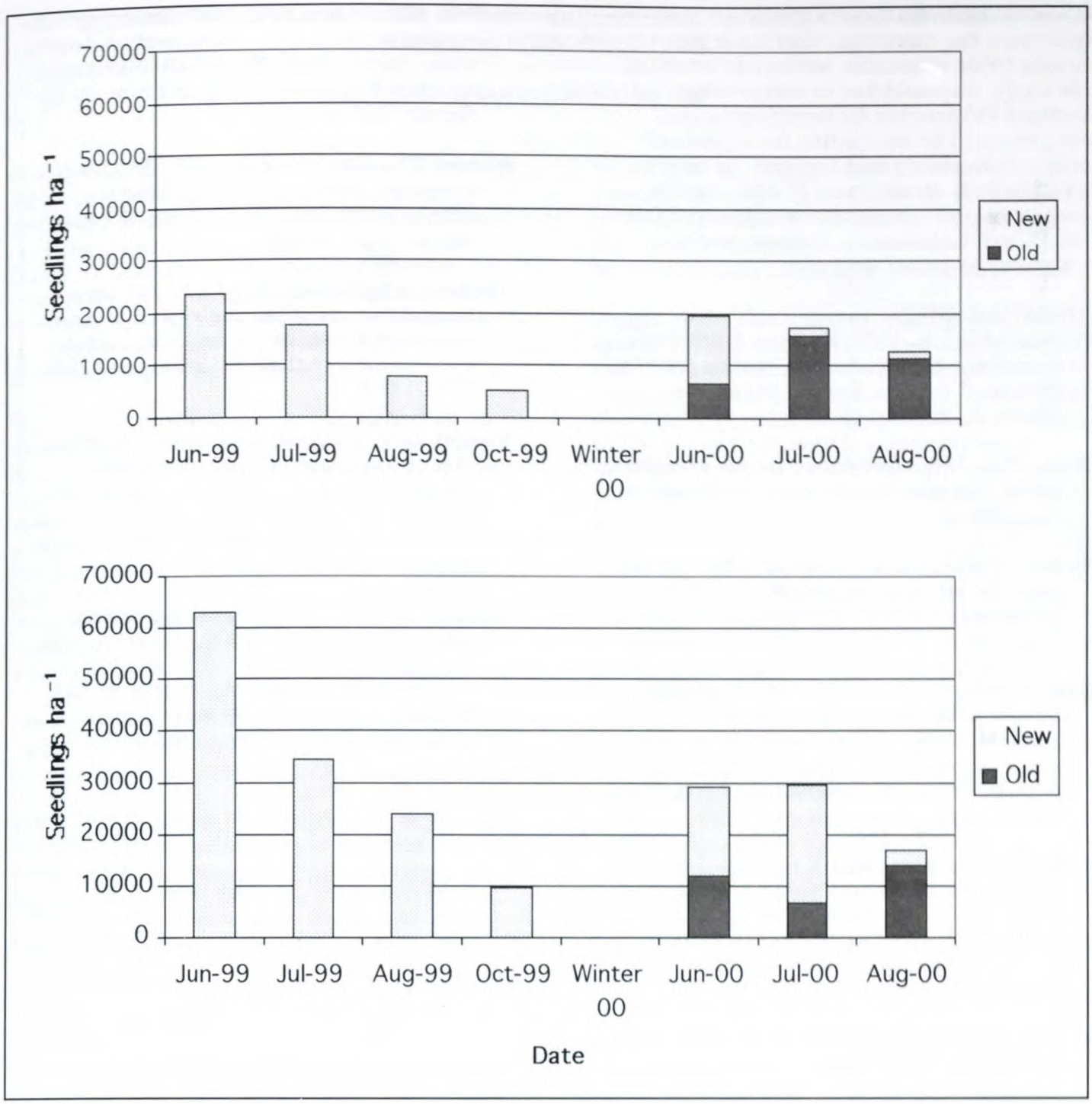

Figure 6.—Number of pin cherry seedlings by age over two growing seasons on NF×G and NF×NG treatment plots. "New" seedlings are germinants from the growing season in which the measurement took place. "Old" seedlings germinated in during the first growing season. For each treatment, "seedlings per hectare" is an average of eight 400 m² treatment plots. Treatment plots are located in four stands on the Westvaco Research Forest in Randolph County, West Virginia.

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MORTALITY TRENDS AND TRAITS OF HARDWOOD ADVANCE REGENERATION FOLLOWING SEASONAL PRESCRIBED FIRES

Patrick Brose and David Van Lear[1]

Fire ecology studies in eastern hardwood forests generally use traditional, plot-based inventory methods and focus on sprouting stems to detect changes in vegetative composition and structure. Fire intensity often is not quantified or even subjectively classified and, if quantified, is not used in subsequent analysis. Consequently, reported responses of hardwood regeneration to prescribed fires vary widely.

To better understand the varying responses among regeneration of common hardwoods to fire, an individual stem mortality approach was employed. Six hundred stems of eight different species were tagged before winter, spring, and summer prescribed burns conducted in 1995. Fire intensity was classified into four categories (low, medium-low, medium-high, and high) based on fuel consumption and fire behavior. Stems not sprouting following topkill were examined for possible causes of mortality.

Among species, hickory (*Carya* spp.) and five upland oak species including black (*Quercus velutina* Lam.), chestnut (*Q. prinus* L.), northern red (*Q. rubra* L.), scarlet (*Q. coccinea* Muenchh.), and white (*Q. alba* L.) were the most resistant to fire with < 15 percent of all stems failing to sprout following topkill. Mortality for these species was highest following moderate- to high-intensity summer fires. Red maple (*Acer rubrum* L.) exhibited little mortality following winter fires but experienced moderate to heavy mortality after spring and summer fires, especially as fire intensity increased. Tulip-poplar (*Liriodendron tulipifera* L.) had moderate to high levels of mortality regardless of season-of-burn.

Tagged stems that failed to sprout were examined for common traits that may indicate why they were susceptible to complete kill while others of the same species were not. Root collar diameter, root collar depth (location in relation to the soil surface), and fire intensity were strongly correlated with rootstock survival and subsequent sprouting while height and previous growth were not. Generally, stems of all species whose root collar was < 0.10 inch diameter and located at the soil surface were completely killed by winter fire regardless of fire intensity. Root collar diameter and root collar depth had to increase to >0.20 and >0.25 inches, respectively, as fire intensity increased and season-of-burn changed to spring or summer.

Our findings indicate why oak and hickory, with their seed burial, hypogeal germination, and root-development strategies are more likely to survive surface fires than many of their competitors. Also, root collar diameter and depth need to be considered during regeneration inventories and before prescribing fire as a silvicultural treatment.

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UNDERSTORY RESPONSE TO DISTURBANCE: AN INVESTIGATION OF PRESCRIBED BURNING AND UNDERSTORY THINNING TREATMENTS

Benjamin J. Dolan and George R. Parker[1]

ABSTRACT.—Lack of disturbance in the Central Hardwood Region has caused a decrease in abundance of shade-intolerant species, such as oaks (*Quercus* spp.) and hickories (*Carya* spp.), in the forest understory, while shade-tolerant species have proliferated. The goal of this research is to determine how two disturbances, prescribed fire and mechanical understory removal, affect woody species regeneration, as well as herbaceous species diversity. A randomized complete block design was developed to test the effect of prescribed burning, mechanical removal of understory shrubs and trees, the combination of burning and removal, and no treatment. Preliminary analysis indicates that all treatments have little effect on herbaceous species diversity; however, prescribed burning affected the composition of woody species seedlings. While fire top-killed most seedlings, regardless of shade tolerance, shade-intolerant species responded by resprouting. Combined with a reduction in the number of shade-tolerant species, burning produced greater equitability among tolerant and intolerant species seedlings. Removal treatments had no effect on the number of seedlings, but increased the level of photosynthetically active radiation (PAR) reaching the forest floor, which is essential for the growth of shade-intolerant species. The combination treatment provides both greater equitability among species, and higher levels of PAR. Further data collection and analysis are necessary to determine if growth and equitability are sustained over time.

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SPATIAL AND TEMPORAL DISTRIBUTION OF FIRE TEMPERATURES FROM PRESCRIBED FIRES IN THE MIXED OAK FORESTS OF SOUTHERN OHIO

Louis R. Iverson, Daniel Yaussy, Joanne Rebbeck, Todd Hutchinson, Robert Long, Brian McCarthy, Cynthia Riccardi, and Anantha Prasad[1]

Prescribed surface fires are being investigated, in conjunction with thinning, as silvicultural tools for assisting in the regeneration of mixed oak forests in the Central Hardwoods Region. Fires were conducted on 2001 March 28 and 2001 April 4-5, respectively, at the Tar Hollow (TAR) and Zaleski (ZAL) State Forests, and at the Raccoon Ecological Management Area (REMA] in southern Ohio. The three sites had been previously surveyed into a 50-m grid. A total of over 400 grid points was established across the three areas.

Each of the sites was previously divided into two treatments of roughly 20 ha each: burn only [B] or thin plus burn [TB]. Thinning had been performed during the 4 months prior to the burns. Just prior to the fires, the grid points were instrumented with stainless steel temperature probes (Type K thermocouple) installed 25 cm above the ground surface, with air temperature logged every 2 s. Buried Hobo®. (Onset Computer Corporation) data loggers successfully captured temperature data for 290 of the points. Because of difficulties with sensor failure at the first burn (TAR), only about a quarter of the grid points were assayed for that site. Nearly 100 percent of the sensors recorded data on the other two sites.

With these data, four measures of fire intensity or timing were available at each grid point:

1) maximum temperature;
2) duration of temperature above 30º C;
3) a heat index, defined as the summed temperatures above 30º C (an integral under the temperature curve); and
4) time of maximum temperature.

These data showed that TAR, though less well assayed because small sample size, had the coolest and most patchy fires, with maximum and average temperatures of 226º C and 103º C for the TB unit and 293º C and 114º C for the B unit (table 1). Less than 30 percent of these sensors recorded temperatures of at least 150º C, a temperature shown earlier to cause significant mortality to saplings.

The REMA fires had maximum and average temperatures of 356º C and 124º C for the TB unit and 354º C and 155º C for the B unit (table 1). A greater percentage (33 to 42 percent) of these sensors recorded temperatures of at least 150º C. The most intense fires were recorded on the ZAL site, with maximum and average temperatures of 415º C and 139º C for the TB unit and 397º C and 172º C for the B unit. The highest proportion (35 to 68 percent) of these sensors recorded temperatures of at least 150º C.

Interestingly, the TB units consistently burned cooler than the B units; perhaps the additional slash resulting from the thinning had not dried sufficiently to increase fire intensity on the sites, but rather it inhibited the movement of flames across the units. In addition, the skid roads and other logging activity likely disrupted the fuel layer, and thus fire movement, in some regions.

The temporal assessment of the maximum temperatures allowed us to evaluate and visualize some aspects of the fire behavior, including a web-based simulation of the fire at the Zaleski site: http://www.fs.fed.us/ne/delaware/4153/ffs/zaleski_burn.html. The simulation shows that fires were generally cooler with a slower rate of spread in the valleys, with hotter, faster fires on the higher ground.

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Table 1.—*Summary of fire temperature data at grid points, at three sites*

Site	Treat[1]	Number Sensors	Max. Temp., °C Abs. Max.	Max. Temp., °C Average Max.	Heat Index[2] Abs. Max.	Heat Index[2] Average Max.	Heat Dur., s Abs. Max.	Heat Dur., s Average Max.	Sensors > 150°C Number	Sensors > 150°C Percent
REM	TB	64	356	124	49880	9865	1007	488	21	32.8
REM	B	65	354	155	42491	12201	1918	971	27	41.5
TAR	TB	18	226	103	22138	5748	1214	363	4	22.2
TAR	B	17	293	114	15364	5205	796	317	5	29.4
ZAL	TB	66	415	139	58625	9968	2956	640	23	34.8
ZAL	B	60	397	172	44029	10050	4472	565	41	68.3
Total/average		**290**	**340**	**134**	**38754**	**8839**	**2061**	**557**	**20**	**38**

[1]B=Burn only, TB=thin and burn

[2]index = temp over 30°C summed over duration (integral under curve)

The spatial assessment of temperature data, when combined with GIS information on vegetation, fuels, topography, weather, and an integrated moisture index, allows us to evaluate some of the factors associated with the fire temperatures. For example, on the ZAL site, there was a slight negative relationship between integrated moisture index and the heat index or duration of burn, i.e., as the topographic moisture regime increased, the fire intensity decreased. Armed with these relationships, we are attempting to model the fire behavior so that spatial estimates of fire temperature and subsequently, fire effects can be achieved.

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COMPARISON OF A LOW-TECH VS. A HIGH-TECH METHOD TO EVALUATE SURFACE FIRE TEMPERATURES

Daniel Yaussy, Joanne Rebbeck, Louis Iverson, Todd Hutchinson, and Robert Long[1]

Prescribed surface fires were conducted in late March-early April 2001, at the Raccoon Ecological Management Area (two ~20 ha areas), and the Tar Hollow (~40 ha) and Zaleski (~40 ha) State Forests in thinned and unthinned mixed-oak forests of southeastern Ohio. Fires are being investigated as a silvicultural tool to aid in regenerating oaks, by removing understory saplings of fire-sensitive species (primarily maples). Previous research has shown that maximum fire temperature is a good variable in evaluating effects and achieving the desired level of topkill.

Tempil, Incorporated, manufactures a series of Tempilaq® temperature sensitive paints, each designed to melt and change color at specific temperatures. This provides a low-cost, "low-tech" method of measuring the maximum temperature achieved at selected locations within the prescribed fires. The visual interpretation of whether paint has melted introduces variability to the estimated maximum temperature.

Onset Computer Corporation produces a variety of miniature data loggers and thermocouples that can be programmed to record temperatures at regular time intervals. This "high-tech" instrumentation costs significantly more than the temperature sensitive paints, but records the time of fire arrival, temperature curve, and duration of heat dissipation.

The sites were instrumented with both low tech and high tech sensors to evaluate air temperatures attained during the fires at more than 400 points arranged on a 50-m grid. Tempilaq® paints that melted at 79°, 121°, 163°, 204°, 315°, and 427° C (175°, 250°, 325°, 400°, 600°, and 800° F) were applied to steel tags that were suspended at 25 cm above the forest floor in conjunction with the thermocouples and buried data recorders. Maximum temperatures recorded by the paints were adjusted to the midpoint between the highest melted paint and the next higher unmelted paint temperature.

A simple linear regression using the maximum temperatures from the paint tags as the independent variable resulted in an r^2 value of 0.84 and MSE of 116.9. There were no differences due to study areas or treatments. The maximum temperatures recorded by the paints sensitive to the highest temperatures were higher than those recorded by the thermocouples. This may be due to human error in interpretation or a true discrepancy. The temperature sensitive paints consistently, and accurately record the maximum temperatures produced by the prescribed fires used by forest managers. In addition to the maximum temperatures produced, the thermocouples and data loggers record residence time, which is needed to calculate fire intensity.

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STAND COMPOSITIONAL DYNAMICS IN A MATURE ILLINOIS OZARKS FOREST: IMPLICATIONS FOR MANAGEMENT

John W. Groninger, Trevor B. Ozier, and Charles M. Ruffner[1]

ABSTRACT.—The Ozark Hills region of southern Illinois is characterized by the severe topography typical of the Ozark Plateau, but is overlain by loess deposits and therefore supports site conditions more typical of mesophytic forests. As is common in lightly disturbed and undisturbed mesic upland sites throughout this region, mature stands in Trail of Tears State Forest are characterized by an oak and hickory-dominated overstory and an under- and mid-story where sugar maple and American beech predominate. Between 1980 and 2000, oak and hickory species generally declined in importance while sugar maple, American beech, and yellow-poplar increased, a trend consistent with those reported in other southern Illinois forests of similar species composition. However, the sugar maple and American beech component at Trail of Tears State Forest are larger and expanding more rapidly than those reported elsewhere within the region. Management options to address these changes are discussed.

Continuously forested public lands in the lower Midwest commonly exhibit a land-use pattern where grazing, occasional burning, and partial cutting have given way to exclusion of all non-recreational activities. These stands commonly feature an overstory dominated by oaks and an understory dominated by shade tolerant mesophytic tree species (Schlesinger 1976, Shotola and others 1992, Zaczek and others 1999).

Trail of Tears State Forest is located in one of the largest blocks of contiguous forest in the lower Midwest. Superficially, most of the forest resembles a maturing-to-old growth, undisturbed stand because much of the forest has been under continuous forest cover and subjected to only partial cutting through the 1980s. A Technical Forestry Plan prepared for the Illinois Department of Conservation in 1984 called for management through selective harvesting. All aspects of the plan involving overstory manipulation have been curtailed since 1989. The availability of data from permanent plot inventories collected in 1980 and again in 2000 provides a unique opportunity to evaluate 20 years of changes across much of this mature oak-dominated forest following cessation of harvesting and burning disturbances.

The objective of this study is to describe changes in tree species composition and dominance within the context of developing guidelines to manage Trail of Tears State Forest.

METHODS

Trail of Tears State Forest encompasses 2,089 ha of the Ozark Hills in Union County, IL. Roughly 93 percent of the terrain is characterized by long narrow ridges dropping off sharply on either side 45 to 60 m to the bottomlands of ephemeral streams, which constitute the remaining 7 percent of the landscape. Slopes are typically steep, restricting roads to ridgetops. Soils are predominantly well-drained silt loams and cherty silt loams.

Most overstory trees originated during one of two periods of recruitment; the 1850s (primarily oak species) and the 1930s (primarily oaks and maples in the absence of fire) (Van de Gevel and others 2002). Since coming under state control in 1929, Trail of Tears State Forest has been

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actively managed for multiple objectives including that of maintaining full stocking with healthy trees. This has involved in part, fire suppression, exclusion of cattle, and partial cutting during the first 55 years of public ownership. A total of 1,017,400 board feet of sawtimber and 1,364 cords of pulpwood were harvested from the forest between 1930 and 1980. No overstory manipulation has occurred on the forest since 1989.

Data Collection and Analysis

Trail of Tears State Forest was inventoried in 1980 using a permanent plot system where 809 m^2 plots were placed across the forest on a 281 m grid for a total of 238 plots. Within these plots, all trees larger than 7.61 cm dbh were identified by species and their diameter measured. These plots were again inventoried in 2000. Plots containing planted pines (n=9) and those subjected to partial cutting following the 1980 inventory (n=38) were excluded from the data presented here (n=191). During the 2000 inventory, four 9 m^2 circular regeneration plots each were established at cardinal directions 7.93 m from the center of each permanent overstory plot. Within these plots, individuals of arborescent species were identified by species and those <7.62 cm dbh but ≥1 m tall were classified as saplings. Stems <1 m in height were classified as seedlings.

Stocking was calculated using standards developed for even aged upland central hardwood stands (Roach and Gingrich 1968). Significance of changes in density and basal area between 1980 and 2000 inventories were compared using paired t-tests (Pallardy and others 1988)

RESULTS AND DISCUSSION

Over the 20 year period of this study, basal area and mean diameter have increased while stem density has decreased (table 1). Overall, stand stocking has increased reflecting a fully stocked condition (Roach and Gingrich 1968).

The trend of decreasing density and increasing basal area is typified by white oak (*Quercus alba* L.) and northern red oak (*Q. rubra* L.). Dominant individuals of these species will be attaining their maximum ages during the next several decades (Loehle 1988, Van de Gevel and others 2003). Canopy disturbances more severe than have been experienced since the onset of public ownership will be required to maintain these species as a significant component of this forest. The typically shorter-lived sassafras (*Sassafras albidum* (Nutt.) Ness) is also following a similar pattern. As is the case with the oaks, the continued presence of sassafras appears to be dependent upon the occurrence of at least partial canopy disturbance. At Kaskaskia Woods, sassafras declined from its

Table 1.—*Summary of stand characteristic changes over 20 years at Trail of Tears State Forest, Illinois*

Stand Characteristic	1980 Inventory	2000 Inventory	Percent change
Overstory density (stems/ha)	505	468	-7.3
Basal Area (m^2/ha)	21.3	24.6	+15.5
Quadratic mean diameter (cm)	23.2	25.9	+11.6
Stocking (percent)	83	93	+12.0
Regeneration Density (stems/ha)	NA	12,839	NA

status as the fourth most frequently occurring species in 1935 to extirpation by 1983 (Zaczek and others 2002).

In contrast to the oaks, sugar maple (*Acer saccharum* Marsh.), American beech (*Fagus grandifolia* Ehrh.), and yellow-poplar (*Liriodendron tulipifera* L.) are increasing in both density and basal area (table 2). Sugar maple and American beech are widely recognized as capable of regenerating and recruiting under a fairly closed forest canopy (Hibbs 1982, Canham 1985). In contrast, yellow-poplar requires at least a partially open canopy to grow beyond regeneration (Orwig and Abrams 1997). Increased density of this species may in part reflect the increasing presence of canopy gaps, through the death of canopy dominant and super dominant individuals, particularly on the lower slope and bottomland positions.

Black oak (*Quercus velutina* Lam.), other oak species, and flowering dogwood (*Cornus florida* L.) showed significant declines in both density and basal area (table 2). The decline of black oak is consistent with the fact that individuals in the 1850s cohort are approaching the recognized maximum age of this species (Sander 1990). Decreased black oak density and basal area in this stand will likely continue, as younger individuals do not replace the senescent cohort.

The decrease in flowering dogwood density and dominance may be at least in part attributed to the emergence of dogwood anthracnose (*Discula destructiva*) as a major cause of decline in the species over the past 2 decades (Carr and Banas 2000). Other factors may also be contributing to decreased dogwood importance as is evidenced by a decline in dogwood density and basal area at Kaskaskia Woods that began decades before

Table 2.—*Density and basal area changes over 20 years among major tree species at Trail of Tears State Forest, Illinois*

	Density (stems/ha)				Basal Area (m^2/ha)			
Species	**1980**	**2000**	**Change**	**p-value**	**1980**	**2000**	**Change**	**p-value**
Acer rubrum L.	10.2	7.6	-2.6	0.08	0.2	0.2	0	0.53
Acer saccharum Marsh.	60.0	102.7	+42.8	0.00	1.1	2.1	+1.0	0.00
Carya spp.	67.7	55.9	-11.8	0.00	2.4	2.5	+0.1	0.56
Cornus florida L.	25.9	21.7	-4.1	0.04	0.2	0.1	-0.1	0.00
Fagus grandifolia Ehrh.	20.0	46.3	+26.3	0.00	0.9	1.5	+0.6	0.00
Fraxinus spp.	21.1	13.6	-7.4	0.00	0.5	0.5	0	0.56
Liquidambar styraciflua L.	8.7	6.0	-2.8	0.04	0.3	0.3	0	0.36
Liriodendron tulipifera L.	12.8	15.0	+2.2	0.04	1.2	2.1	+0.9	0.00
Nyssa sylvatica Marsh.	9.3	11.9	+2.7	0.03	0.2	0.3	+0.1	0.06
Ostrya virginiana (Mill.)K Koch	17.5	17.2	-0.3	0.85	0.2	0.1	-0.1	0.16
other non-*Quercus*[1]	13.5	9.4	-4.1	0.00	0.4	0.3	-0.1	0.09
Quercus alba L.	99.4	72.5	-26.8	0.00	5.4	6.8	+1.3	0.00
Quercus rubra L.	41.9	24.0	-17.9	0.00	2.6	3.1	+0.5	0.02
Quercus velutina Lam.	36.3	18.2	-18.0	0.00	4.7	3.5	-1.2	0.00
other *Quercus* spp.[2]	6.5	2.0	-4.5	0.00	0.4	0.2	-0.2	0.03
Sassafras albidum (Nutt.)Ness	47.9	37.7	-10.2	0.00	0.6	0.7	+0.1	0.01
Ulmus spp.	6.7	5.8	-0.9	0.34	0.4	0.2	-0.2	0.84

[1] Includes *Amelanchier arborea* (Michx.) Fern., *Juglans nigra* L., *Magnolia acuminata* L., *Platanus occidentalis* L., *Prunus serotina* Ehrh., and 16 additional species.
[2] Includes *Quercus bicolor* Willd., *Q. coccinea* Muenchh., *Q. macrocarpa* Michx., *Q. michauxii* Nutt., *Q. muhlenbergii* Engelm., *Q. prinus* L., and *Q. stellata* Wangenh.

the first documented occurrence of anthracnose in Illinois (Schwegman and others 1998, Zaczek and others 2002). Perhaps a similar decline will occur in the Trail of Tears State Forest as well.

The seedling stratum is dominated by intermediate to tolerant tree species including sugar maple, red maple (*Acer rubrum* L.), hickory (*Carya* Nutt. spp.), white ash (*Fraxinus americana* L.), and hophornbeam (*Ostrya virginiana* (Mill.) K.Koch) (table 3). Other important understory and mid-story species in the seedling and sapling strata include paw paw (*Asimina triloba* (L.) Dunal) and spicebush (*Lindera benzoin* (L.) Blume). Although many other species, including oaks, are present in the understory, only sugar maple and American beech are consistently recruiting into the sapling size class and beyond. These results are consistent with observations at Kaskaskia Woods where no oak recruitment has occurred in 62 years of observation (Zaczek and others 2002). Similarly, few oak saplings were recorded at Weaver Woods (Shotola and others 1992).

Our results suggest that the low intensity of disturbances at Trail of Tears State Forest is permitting the development of only shade tolerant species. Therefore, the decline of oaks and other disturbance dependent species and the emergence of sugar maple and American beech is likely to continue as long as a management

Table 3.—*Regeneration composition of Trail of Tears State Forest, Illinois during summer 2000*

Species	Seedlings (< 1 m height)		Saplings (>1 m height and <7.61 cm dbh)	
	Density (stems/ha)	Relative Density	Density (stems/ha)	Relative Density
Acer rubrum	908	7.9	23	1.7
Acer saccharum	744	6.5	279	21.4
Asimina triloba	2225	19.3	44	3.3
Carya spp.	637	5.5	48	3.7
Cornus florida	99	0.9	59	4.5
Fagus grandifolia	811	7.0	508	38.8
Fraxinus spp.	899	7.8	13	1.0
Lindera benzoin	445	3.9	11	0.9
Liriodendron tulipifera	155	1.3	24	1.8
Nyssa sylvatica	154	1.3	14	1.1
Ostrya virginiana	684	5.9	92	7.0
other non-*Quercus*	3,223	28.0	169	12.9
Quercus alba	461	4.0	10	0.8
Quercus rubra	162	1.4	0	0
Quercus velutina	458	4.0	1	0.1
other *Quercus* spp.	42	0.4	0	0
Sassafras albidum	1,654	14	44	3.3
Ulmus spp.	440	3.8	24	1.8
Total	11,531	100	1,308	100

program characterized by little or no canopy disturbance is pursued (Trimble 1965, Schlesinger 1976).

Similarities among Trail of Tears State Forest and the less disturbed Kaskaskia Woods and Weaver Woods are striking despite differences in the disturbance regimes (table 4). Kaskaskia Woods was last partially cut during the 1910s (Schlesinger 1976). Fire and grazing have been excluded since 1933. Weaver Woods was free of grazing and subjected to removal of fewer than 100 trees over 7.2 ha from 1871 until 1950 when all cutting ceased (Weaver and Ashby 1971).

The higher density and basal area of shade tolerant sugar maple and American beech at Trail of Tears State Forest may be attributed to the fact that this forest is in an earlier stage of stand development than Kaskaskia Woods and Weaver Woods. In Kaskaskia Woods, the rate of sugar maple density increase has slowed somewhat since the 1950s. Alternately, the use of partial cutting at Trail of Tears State Forest without subsequent control of the shade tolerant mid-story may be contributing to increased growth of maple and beech. Partial cutting regimes may in effect simulate gap phase regeneration and recruitment dynamics displayed by these species (McGee and others 1999), potentially hastening the conversion to late successional species (Abrams and Scott 1989) if midstory control measures are not implemented.

Inherent site quality differences among the three stands discussed here may play a role in species composition and growth patterns. Trail of Tears State Forest is western-most along a gradient of surficial loess deposits that decrease from West to East. This, along with the presence of bottomland sites may be responsible for the rapid rate of increase in sugar maple and yellow-poplar basal area exhibited in the Ozark Hills.

MANAGEMENT IMPLICATIONS

The results presented here suggest that restricting overstory manipulation to partial cutting has produced stand conditions similar to those observed in forests where cutting has been curtailed for several decades. Regeneration and recruitment of overstory tree species, although abundant, is restricted to a small number of shade tolerant species. Failure to accomodate the regeneration and recruitment requirements of less shade tolerant species will, in the absence of natural catastrophic disturbance, result in their decline and likely eventual extirpation.

Some form of canopy disturbance will likely be necessary to secure sufficient regeneration of current canopy dominant species. These might include the use of shelterwood or two age systems, including the use of prescribed fire (Moser

Table 4.—*Annualized percent changes among major overstory tree species density and basal area in three southern Illinois forests*

	Trail of Tears State Forest		Weaver Woods[1]		Kaskaskia Woods[2]	
Species	**Density**	**Basal Area**	**Density**	**Basal Area**	**Density**	**Basal Area**
Acer saccharum	+3.6	+4.5	-0.2	+2.6	-0.4	+3.4
Carya spp.	-0.9	+0.2	-1.5	+0.3	-1.8	+0.1
Fagus grandifolia	+6.6	+3.3	-0.7	+0.2	NA	NA
Liriodendron tulipifera	+0.9	+3.8	-0.3	+0.9	+0.3	+2.1
Quercus alba	-1.4	+1.3	-0.5	+1.7	-2.2	+0.1
Quercus rubra	-2.2	+1.0	-1.3	0.0	-2.3	-0.6
Quercus velutina	-2.5	-1.3	-1.8	-1.5	-1.9	-0.5

[1] Adapted from Shotola and others (1992).
[2] Adapted from Zaczek and others (2002).

and others 1996, Brose and others 1999, Ruffner and Davis 2001). Adjustment of a single tree selection regime (Loewenstein and others 2000) where shade tolerant mid-story and understory species are controlled (Kochenderfer and others 2001) may also accomplish this objective.

If a system dependent upon sustained partial cutting is implemented, care must be taken to assure that volume of trees removed is balanced by ingrowth of the same species so that a diverse overstory is maintained. If harvesting were continued at Trail of Tears State Forest in such a manner that oak regeneration is not secured, sugar maple and American beech ingrowth could produce a sustained flow of volume while resulting in a conversion to non-oak dominance. To date, management systems with relatively low visual impact have failed to secure regeneration of the dominant oaks and hickories, yet have also aroused sufficient public outrage to suspend overstory manipulation indefinitely. The more drastic disturbances required to secure a new cohort of oak and hickory species will be extremely challenging to implement in the current social and political environment.

ACKNOWLEDGMENT

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SAWTIMBER OAK STAND RESPONSE TO SIX DISTINCT CUTTING METHODS

Jeffrey S. Ward and George R. Stephens[1]

Nonindustrial private owners control over 80 percent of the 50 million acres of oak-dominated forest in the northern United States. Although non-commodity amenities are more important than timber production for many of these owners, the expense of forest ownership may make them amenable to forest management practices that provide income while retaining the desired non-commodity amenities. The effects of distinct cutting methods on stand growth and dynamics were examined in a study established in 1981 to 1984 in three southern New England forests. Cutting methods were: shelterwood, diameter limit, coppice with standards, commercial clearcut, silvicultural clearcut, and an unmanaged control. The cutting method plots (4 to 7 acres) were on medium quality sites (SI = 65). Permanent 10-factor prism plots were established before initial treatment, were remeasured in 1998-99, and again in 2001 following the second cutting cycle.

Basal area averaged 92 ft^2/ac before harvest, 68 percent of which was in sawtimber trees. The initial harvests reduced basal area by: diameter limit (38 percent), shelterwood (56 percent), coppice with standards (58 percent), and commercial clearcut (76 percent). Over the next 16 to 17 years, annual basal area growth was highest on commercial clearcut plots (2.8 ft^2/ac) and lowest on shelterwood plots (1.5 ft^2/ac). Diameter limit and commercial clearcutting accelerated the shift from oak-dominated to maple/birch-dominated forests. The coppice with standards treatment retained oak in both the poletimber and sawtimber size classes. Cubic-foot volume growth did not differ significantly among cutting methods, averaging 56 ft^3/ac/yr (fig. 1).

After the second cutting cycle, the proportion of poletimber volume (ft^3/yr) lost to cull was highest in the commercial clearcut plots (13 percent). The other partially cut plots, diameter limit (5 percent), and coppice with standards (4 percent), had lower cull losses than the unmanaged control plots (8 percent). Absolute cull losses were also higher in the commercial clearcut plots (fig. 2).

Board-foot volumes (1/4-inch International) averaged 8,200 bf/ac before the initial harvest. Sixty-eight percent of sawtimber trees had butt-log grades of 2 or better. Harvested volumes ranged from 3,400 bf/ac on the coppice with standards plots to 7,800 bf/ac on the commercial clearcut plots. Somewhat surprisingly, board-foot volume growth was similar on unmanaged, diameter limit, and shelterwood, and coppice with standards plots (259 bf/ac/yr). Volume growth was significantly lower on commercial clearcut plots (118 bf/ac/yr). Total board-foot yields (final volumes plus harvested volumes) for the silvicultural clearcut (7,400 bf/ac) and commercial clearcut plots (10,100 bf/ac) were significantly lower than for uncut, shelterwood, and diameter limit cuts, 12,800, 13,100, and 13,500 bf/ac, respectively (fig. 3).

This study found that four distinct treatments: shelterwood, diameter limit, coppice with standards, and no management resulted in similar stand volume (board foot) growth rates. Therefore, the first three methods can be used by landowners who wish to generate income to offset expenses (real estate taxes, insurance, management costs) and realize a return on investment. The choice among cutting methods should be dependent on the esthetic and regeneration goals of the landowner. Although commercial clearcutting had an initially higher harvested volume, low quality of residual trees and depressed stand growth rates indicate it is not a viable option for long-term forest management.

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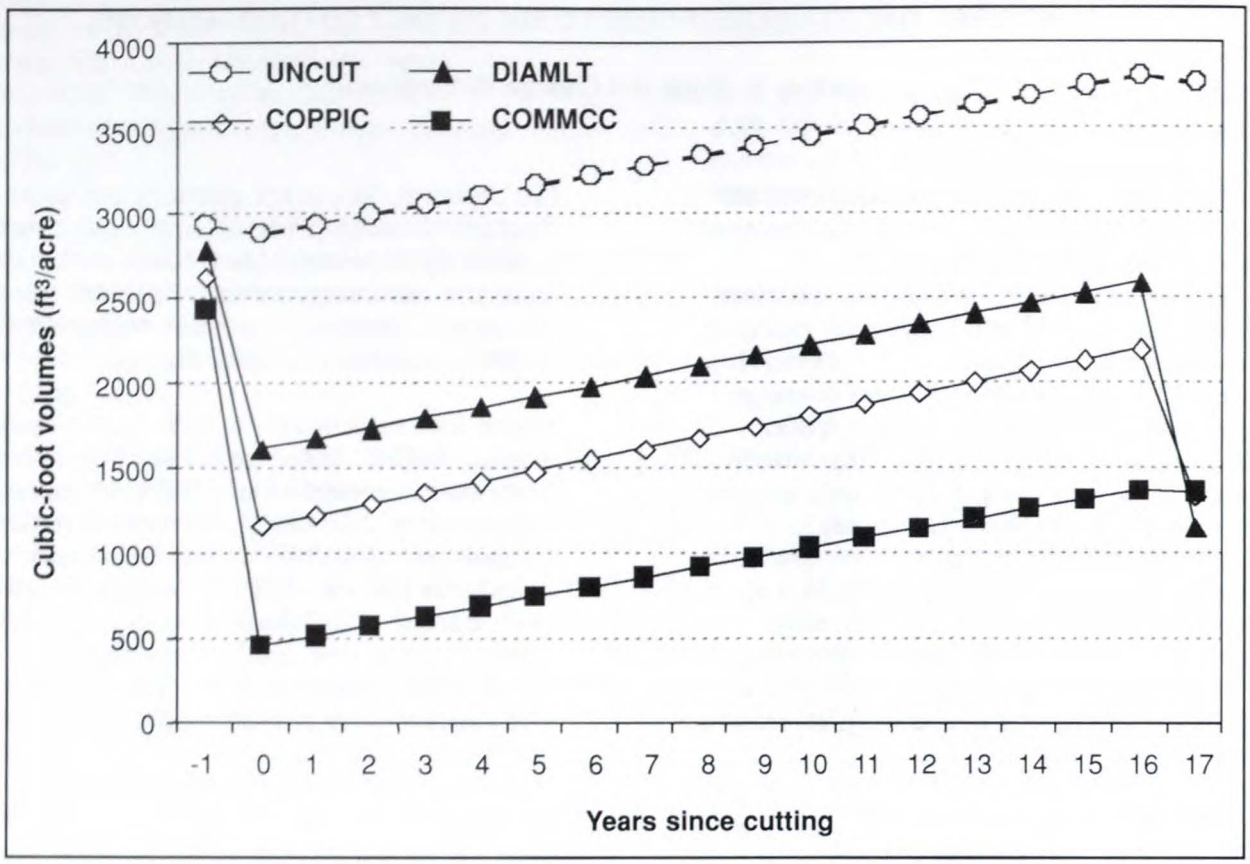

Figure 1.—*Changes in cubic-foot volume (ft³/ac) over time by cutting method. Stands were cut in years 0 and 17.*

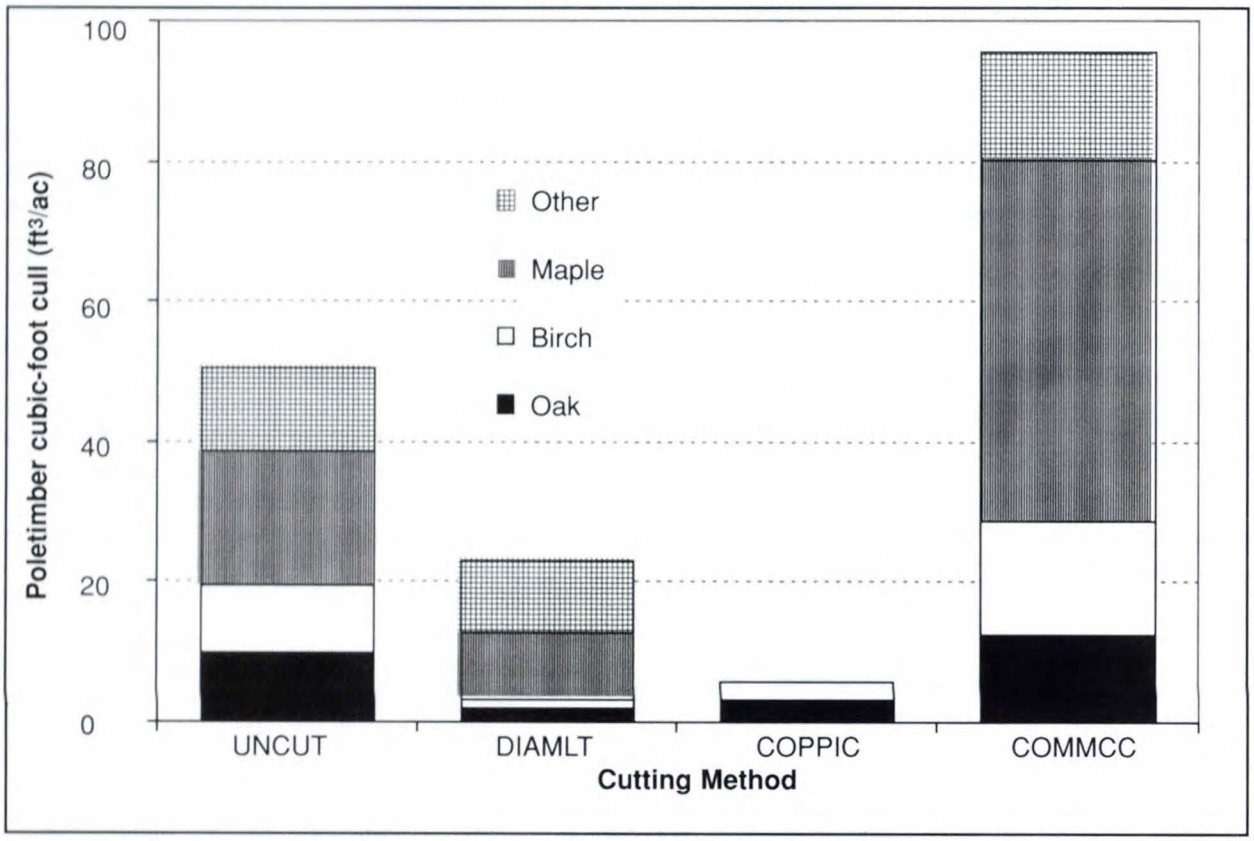

Figure 2.—*Cull volume loss (ft³/ac) in residual poletimber after the second cutting cycle by cutting method and species group.*

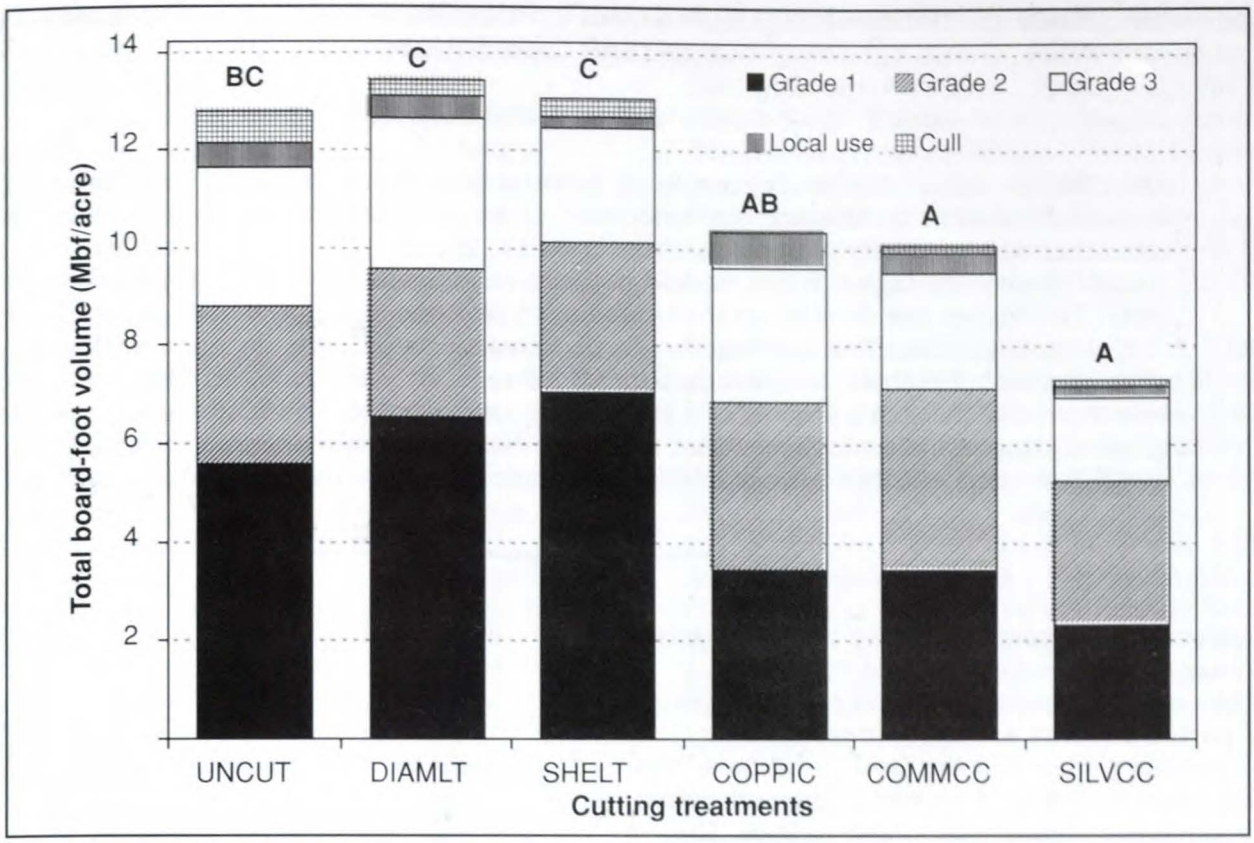

Figure 3.—*Total board-foot volume (International 1/4 Mbf/ac) 14-17 years after initial harvest by cutting method. Total volume = harvested + residual + growth. Total volumes with the same letter were not significantly different at $p \leq 0.05$.*

INFLUENCE OF THINNING STYLE ON STAND STRUCTURE AND GROWTH IN UPLAND OAKS: A 58-YEAR CASE STUDY

Jeffrey S. Ward[1]

ABSTRACT.—In 1937, a study comparing low and high thinning (partial crop tree release) was established in northwestern Connecticut. Oaks accounted for 65 percent of the crop trees that were partially released at stand ages 17, 26, and 42 years. Sawtimber trees had greater diameters, higher volumes, and higher tree grades on thinned than unmanaged plots. The higher oak density on the unmanaged plot was concentrated in poletimber trees (5 to 10 inches dbh). Tree sapling and shrub densities were higher on the low thinning plots. Density of a shade tolerant midstory dominated by red maple (*Acer rubrum* L.) poles was increased by high thinning, but reduced by low thinning. The minimal gain in tree grade on thinned plots indicates that the high cost of thinning/cleaning in sapling stands, as implemented in this study, is unlikely to be recovered in future harvests of mature sawtimber.

Current management practices have long-term consequences on future forest resources. Uneven-aged management and extended periods of partial cuttings are being prescribed for increasing portions of the forest, and have been advocated as the preferred management option on some national forests (Guldin 1996). The long-term effects of partial cutting have been interpolated from short-term and stand reconstruction studies, because of the paucity of long-term studies. These studies suggest partial cutting accelerates the shift in dominance from midtolerant to tolerant species and may eventually decrease stand growth rates (Lamson and Smith 1991). Thus, changes in current management practices will affect the quality and variety of future forest resources.

Short-term impacts (< 10 years) of partial cutting are well documented. Partial cutting increases growth of residual trees (Hilt 1979, Sonderman 1984, Stringer and others 1988). Medium-term studies (10 to 40 years) suggest the initial gains in volume growth following partial cutting are not lost over time (Ward 1991). Thirty-two years after a single thinning, black and scarlet oak volumes were higher on thinned than control plots (Dwyer and Lowell 1988). Partial cuttings at 5-year intervals over a 20-year period increased black and northern red oak diameter growth (Rudolph and Lemmien 1976). Stand volume growth was higher on partially cut than control plots in West Virginia (Smith and Miller 1987).

Studies in unmanaged forests and research on partial cutting have reported the same pattern of increasing dominance by tolerant species (Rudolph and Lemmien 1976, Christensen 1977, Parker and others 1985, Ward 1992). While partial cutting has been reported to increase oak regeneration density (Kittredge and Ashton 1990), non-oak species often dominate regeneration after partial cutting in upland oak forests (Smith 1979, Hill and Dickman 1988, Heiligmann and others 1985). Stand reconstruction studies suggest partial cutting has accelerated succession towards dominance by more tolerant species (Jokela and Sawtelle 1985, Abrams and Scott 1989). However, heavy partial cutting (> 50 percent canopy removal) may allow northern red oak to achieve eventual canopy dominance (Lorimer 1983).

One method of partial cutting, crop tree management, has been demonstrated to enhance

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growth of selected crop trees and has been advocated as an alternative to traditional area-wide thinning (Perkey and others 1994). Therefore, it is prudent from both economic and ecological perspectives to document the long-term impact of this method of partial cutting on forest structure and composition.

The objective of my research was to examine the influence of two distinct methods of thinning (high and low) on short-term changes in crop tree growth response and long-term changes in stand structure and volume. This research illustrates the consequences of different methods of thinning on long-term changes in species composition and diameter distribution. These results can be used by forest managers to determine whether long-term changes initiated by thinning in young stands are appropriate for their management goals.

STUDY AREA

In 1937, the Connecticut State Forestry Department established a thinning (partial crop tree release) study in the Housatonic State Forest in northwestern Connecticut. Soils are well-drained Charlton-Hollis fine sandy loams (Typic Dystrochrepts) with 0-20 percent slopes derived from schist and granite glacial till (Hill and others 1980). Mean site index (black oak) is 68 feet. The area is in the northern temperate climate zone with 45 inches of annual precipitation evenly distributed over the year. The study plot was repeatedly cut over for charcoal production between 1840-1920, and numerous charcoal mounds are still evident.

Five species groups were found on the plots: Upland oaks—*Quercus alba* L., *Q. coccinea* Muenchh., *Q. prinus* L., *Q. rubra* L., and *Q. velutina* Lam.; Maple—*Acer rubrum* L. and *A. saccharum* Marsh.; Aspen—*Populus grandidentata* Michx.; Other—*Betula lenta* L., *B. papyrifera* Marsh., *Carya glabra* (Mill.) Sweet, *Fagus grandifolia* Ehrh., *Fraxinus americana* L., *Liriodendron tulipifera* L., *Pinus strobus* L., *Sassafras albidum* (Nutt.) Ness, and *Tilia americana* L.; and Shrubs—*Acer pensylvanicum* L., *Amelanchier arborea* (Michx.) Fern., *Carpinus caroliniana* Walt., *Castanea dentata* (Marsh.) Borkh., *Cornus florida* L., *Corylus cornuta* Marsh., *Hamamelis virginiana* L., *Kalmia latifolia* L., and *Ostrya virginiana* (Mill.) K Koch.

Increment cores of three trees/plot at 1 foot aboveground in 1962 determined the stand was established around 1920, i.e., the study was established in a 17-year-old stand. Aspen was the most numerous species in the 1- to 4-inch diameter class at stand age 17 (table 1), accounting for 52 percent of all stems. Upland oak predominated in the ≥ 5 inch diameter class (71 percent of stems). Average density before cutting was 2,360 stems/acre (dbh ≥ 1 inch). Initial basal area was slightly higher on the thinned plots, 43 ft^2/acre, than on the unmanaged plot, 36 ft^2/acre.

METHODS

Selection criteria for crop trees were not stringent, "the best trees, which promised to make the final crop." (Hawes 1955). Crop tree density ranged from 135 stems/acre on low thinning plots to 147 stems/acre on control plots. Oaks accounted for 65 percent of crop trees, hickory 15 percent, maple 10 percent, and other species 10 percent. Slightly more than half of crop trees were in the intermediate crown class prior to the initial thinning. At the beginning of this study, mean diameter of oak crop trees was 3.5 inches and mean height was 32 feet.

Two plots were thinned from below (1.5 acres total), two plots were thinned from above (1.6 acres total), and one plot was left unmanaged to serve as a control (0.9 acres). In the low thinning plots, trees in the intermediate and suppressed crown classes that were competing

Table 1.—*Initial (precut) distribution of stems (n/ac) among diameter classes by species groups at stand age 17 in the Housatonic Blue Ribbon plots, Connecticut*

	Diameter class (inches)					
Species group[1]	1	2	3	4	> 5	Total
Upland oaks	169	187	100	42	12	510
Aspen	352	485	317	67	4	1225
Maple	357	104	11	0	0	472
Other	99	49	4	0	1	153
Total	977	825	432	109	17	2360

[1]Species are listed in Methods.

with crop trees were removed. Shrub species were not cut. In the high thinning plots, trees in the dominant and codominant crown classes that were competing with crop trees were removed. The one remaining tally sheet of stems cut at stand ages 17 indicates that 59 percent of cut stems were < 3 inches dbh and 76 percent of these were either aspen or maple. This roughly corresponds with the composition prior to the initial thinning.

Crop trees were released in 1937, 1946, and 1962 at stand ages 17, 26, and 42, respectively (fig. 1). In addition to the thinnings, several oaks with diameters greater than 10 inches were girdled in 1937 to eliminate potential wolf trees. Thinning reduced basal area by 28, 22, and 13 percent on high thinning plots at stand ages 17, 26, and 42 years. For the same periods, basal area was lowered by 45, 19, and 5 percent on low thinning plots.

Aspen and oak accounted for nearly 95 percent of basal area cut at stand age 42 on high thinning plots (table 2). Maple accounted for over 60 percent of trees cut on low thinning plots. Several pole-sized oaks in front of a deer stand were the only trees cut since 1962.

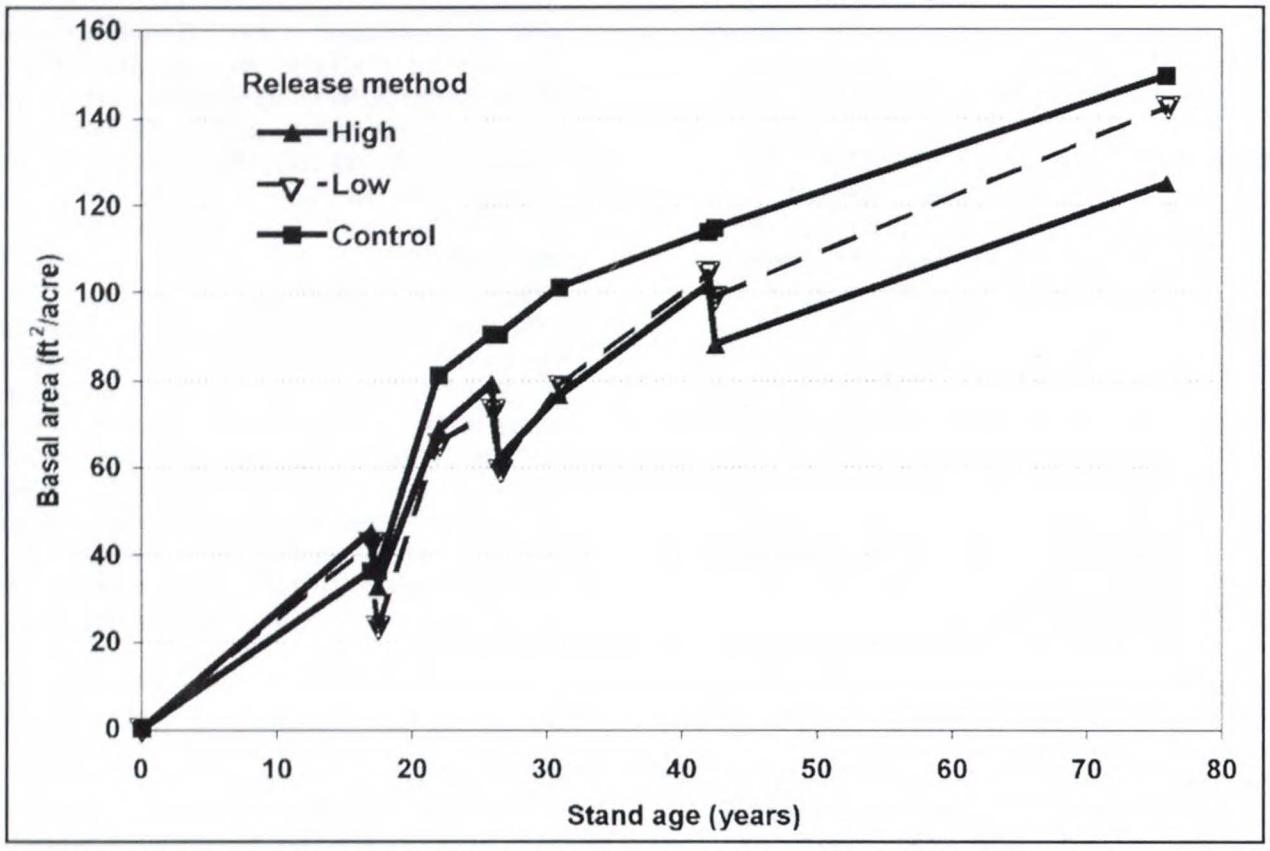

Figure 1.—*Basal area growth (ft²/ac) by release method in Housatonic Blue Ribbon plots, Connecticut. Treatments: High thinning (partial crop tree release from above), Low thinning (partial crop tree release from below), Control (no thinning).*

Table 2.—*Density (n/acre) and basal area (ft²/ac) removed by thinning method (high/low) at stand age 42 in the Housatonic Blue Ribbon plots, Connecticut*

Species group[1]	Density (n/ac)		Basal area (ft²/ac)	
	High	Low	High	Low
Upland oaks	11	12	3.6	1.4
Aspen	28	5	9.7	1.3
Maple	4	94	0.5	3.4
Other	2	36	0.3	1.0
Total	45	147	14.1	7.0

[1]Species are listed in Methods.

High thinning at stand age 17 required 3.7 man-days/acre, low release required 8.0 man-days/acre. The thinnings at stand ages 17 and 26 removed a combined 410 ft^3/ac on these plots. Volume data for trees cut on low thinning plots has been lost. Harvests at stand age 42 removed 330 and 120 ft^3/ac on high and low thinning plots, respectively.

Field Measurements

Selected crop trees on all plots were banded at 4.5 feet and numbered with blue paint. Diameters were recorded to the nearest 0.1 inch. Heights were recorded to the nearest 1 foot for all crop trees. All other trees > 0.5 inches dbh were tallied by species and by 1-in. dbh classes (table 1). Trees that were cut to release crop trees were tallied by species and 1-in. dbh classes. Individual crop tree measurements (diameter, height, and crown class) for 1937, 1942, 1946, and 1951 exist for three of the plots. Individual tree data for the other two plots were lost.

Complete stand summaries by species and crown class were recorded following the 1937, 1942, 1946, and 1951 inventories. Crown class designation standards through the survey in 1962 were based on Hawley and Hawes (1925), which are nearly identical to those used in the 1995 survey (Smith 1962). For this paper, upper canopy is defined as trees in the dominant and codominant crown classes.

Plot corners were permanently marked with stone cairns and chestnut posts. All original plot corners were relocated in 1995 and the original chestnut posts were replaced with iron stakes. Species and dbh were recorded for all stems > 0.5 inches dbh. Diameters at stump height (dsh, 1 foot aboveground) were measured for all trees > 4 inches dbh. Total height, pulp height (to 4 inch tip), and saw log height (to 9 inch tip) were recorded to the nearest foot for all trees ≥ 10.6 inches dbh. Pulp height was recorded for trees with diameters between 4.6 and 10.5 inches. Tree grades were assigned using standards for the northeastern forest survey (Alerich 1996). Stump diameter and species were recorded for all snags and dead and down trees.

Data Analysis

Statistical analysis of stand level characteristics (basal area, board-foot and cubic volume) was not possible because there was no replication of the control plot. Therefore, this research was treated as a case study and data from the 2 high thinning plots were pooled as were those from the 2 low thinning plots. Pearson chi-square statistics were used to determine whether the distribution of stems among species groups were independent of thinning method (high/low/control). Because shrub species were not tallied until the survey in 1995, they were not included in the chi-square analysis. Pearson chi-square statistics were also used to determine whether the distribution of oak sawtimber stems among tree grades were independent of thinning method. Tukey's HSD test was used to test differences of quantitative tree data (dbh, saw log height, pulpwood height, volume) among treatments. Differences were judged significant at $p \leq 0.05$.

Board-foot (International 1/4-inch rule) volumes and cubic-foot volumes were calculated using equations developed by Scott (1979, 1981). Volumes cut in 1937 and 1946 were originally recorded in cords. These were converted to cubic feet using 60 cubic feet per cord (Wenger 1984, p. 277). Trees were categorized by diameter class: saplings (0.6 to 4.5 inches dbh), poles (4.6-10.5 inches dbh), and sawtimber (≥ 10.6 inches dbh).

RESULTS

Basal area steadily increased on all plots after thinning. At stand age 76, lower basal area was observed on high thinning (125 ft^2/ac) and low thinning (143 ft^2/ac) plots, than on the unmanaged plot (150 ft^2/ac). The basal area difference among thinning methods was not related to differential stand basal area growth rates (fig. 1). Basal area growth rates were similar among thinning methods for a given period. Basal area growth decreased with stand age from a high of 5.5 ft^2/ac/yr between stand ages 17 and 26, to 1.6 ft^2/ac/yr between stand ages 26 and 42, to 1.2 ft^2/ac/yr between stand ages 42 and 76.

Crown class transitions of crop trees were not significantly different among species or thinning methods. Therefore, species and thinning methods were pooled (table 3). Nearly all crop trees in the dominant crown class at stand age 17 remained in the upper canopy through stand age 31. In contrast, nearly half of crop trees in the intermediate crown class regressed into the suppressed crown class and fewer than 12 percent were in the upper canopy at stand age 31.

On all plots, there were an average of 16 snags/acre and 39 dead and down trees/acre at stand age 76. The dbh of dead and down trees was predicted using stump diameter (dsh):

Oak: dbh = 0.98 + 0.72*dsh, n=329, r^2=0.95, and Other: dbh = -0.31 + 0.89*dsh, n=584, r^2=0.97. Both absolute and relative oak mortality (snags and dead/down at stand age 76) were higher on control than thinned plots. Oak accounted for 69 percent of the 20 ft^2/ac mortality on the unmanaged plots compared with 45 percent of the 13 ft^2/ac on the high thinning plots. In contrast, aspen accounted for the largest proportion of mortality on the high (46 percent) and low (48 percent) thinning plots.

The distinct thinning methods resulted in distinct diameter distributions by stand age 76 for several species groups (fig. 2). Both high and low thinning reduced the number of smaller oaks, but did not increase the density of larger, more valuable trees. In contrast, both thinning methods increased the number of larger diameter aspen compared with the unmanaged plot. There were less than three aspen stems per acre larger than 13 inches dbh on the unmanaged plot, compared with 20 and 32 per acre on high and low thinning plots, respectively. Relative to the unmanaged plot, high thinning decreased the number of red maple saplings and increased the number of red maple pole trees. The lowest number of red maple saplings was found on the low thinning plots. Shrub density was highest on low thinning and lowest on high thinning plots.

Table 3.—*Crown class transitions between stand ages 17 and 31 of crop tree on the Housatonic Blue Ribbon plots, Connecticut. Only three plots included because of missing data*

Crown class at stand age 17	N[1]	Crown class at stand age 31(percent)			
		Upper canopy	Intermediate	Suppressed	Mortality
Dominant	29	96.6	3.5	0.0	0.0
Codominant	101	76.2	18.8	5.0	0.0
Intermediate	183	11.5	39.3	44.3	4.9

[1]Sample size

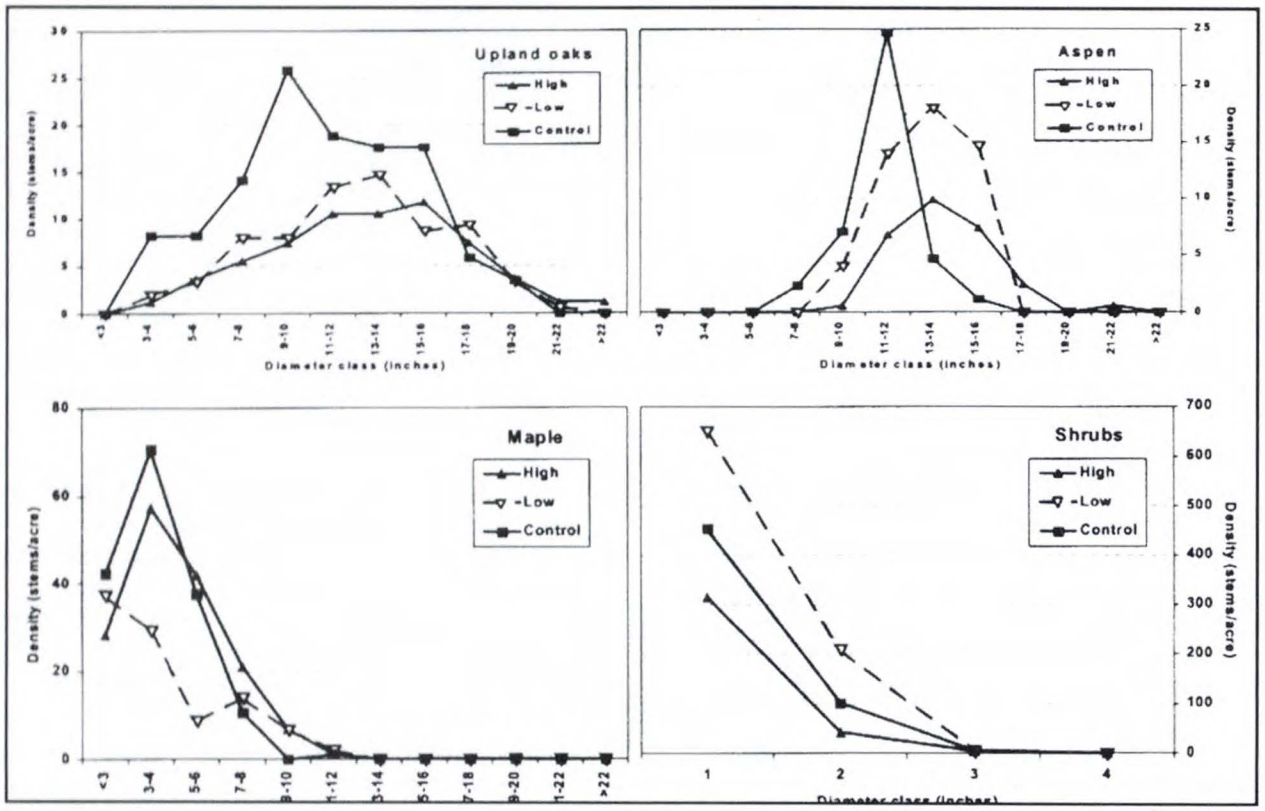

Figure 2.—*Distribution of stems by diameter class and species groups at stand age 76 in Housatonic Blue Ribbon plots, Connecticut. Treatments: High thinning (partial crop tree release from above), Low thinning (partial crop tree release from below), Control (no thinning). Diameter classes for upland oaks and aspen are identical to those for maple.*

Significant differences ($\chi^2 = 121.5$, d.f. = 6, $p \leq 0.001$) in species composition among treatments had developed by stand age 31 (table 4) and remained through stand age 76 ($\chi^2 = 129.8$, d.f. = 6, $p \leq 0.001$). Absolute and relative oak densities were higher on unmanaged than thinned plots. The differences in oak density among the thinning methods increased with stand age. By stand age 76, oak density on the high thinning plots (65 stems/ac) was less than half that on the unmanaged plot (131 stems/acre). However, most of the increased oak density was in the smaller diameter classes (fig. 2).

Maple density on low thinning plots was 42 percent lower than on the unmanaged plot and 69 percent lower than on high thinning plots at stand age 31 (table 5). Relative maple densities increased more on high thinning and control plots than on low thinning plots. Both relative and absolute aspen densities have fallen through stand age 76 on thinned plots (71 to 88 percent). Relative aspen density decreased from over 30 percent at stand age 31 to less than 10 percent at stand age 76. Aspen densities were lower on thinned plots than the unmanaged plot.

Cubic-foot and board-foot volumes were higher on low thinning plots than either the high thinning or unmanaged plots at stand age 76 (table 5). However, stand volumes of the more valuable oak species differed little among thinning methods.

Table 4.—*Density (n/acre) of stems (> 1.5 inches dbh) by species group and stand age for each thinning method on the Housatonic Blue Ribbon plots, Connecticut*

Period	Species group[1]	Thinning method		
		High	Low	None
Stand age 31	Upland oak	272	148	452
	Aspen	254	197	316
	Maple	227	70	121
	Other	91	51	89
	Total	844	466	979
Stand age 42	Upland oak	187	140	301
	Aspen	133	138	139
	Maple	302	133	209
	Other	147	121	105
	Total	769	532	754
Stand age 76	Upland oak	65	79	131
	Aspen	31	56	46
	Maple	165	101	171
	Other	98	247	171
	Total	359	483	519

[1]Species are listed in Methods.

Table 5.—*Stand volumes at stand age 76 by thinning method and species group on the Housatonic Blue Ribbon plots, Connecticut*

Species group[1]	Thinning method		
	High	Low	None
	Board-foot volume (International ¼/ acre)		
Upland oak	5772	5687	6061
Non-oak species	3163	5241	1484
Total	8934	10929	7545
	Cubic-foot volume (ft^3/acre)		
Upland oak sawtimber	1564	1627	1806
Non-oak sawtimber	956	1534	728
Poles	560	604	1093
Total	3080	3765	3626

[1]Species are listed in Methods.

Aspen accounted for most of the "Other" sawtimber volume. Cubic-foot volume of pole trees was much higher on the unmanaged than thinned plots.

There were marginal differences in the characteristics of saw log oak trees at stand age 76 (table 6). Trees on high thinning plots had slightly larger diameters and longer saw logs than trees on the unmanaged plot. These differences resulted in higher per tree volumes on high thinning plots compared with the unmanaged plot. Board-foot volumes were 26 percent higher, and cubic-foot volumes were 17 percent higher on high thinning than unmanaged plots. Significant differences ($\chi^2 = 13.5$, d.f. = 6, $p \leq 0.04$) in tree grade of oak sawtimber among treatments had developed by stand age 76. Both high and low thinning plots had more Grade 1 and 2 oak sawtimber than did the unmanaged plot. A higher proportion of potential Grade 1 oak sawtimber (≥ 16 inches dbh) were Grade 1 on the high and low thinning plots (75 and 69 percent) than on the unmanaged plot (50 percent).

Differences in saw log characteristics among thinning methods were more pronounced for the non-oak saw log trees (table 6). Mean diameter was significantly greater on both high and low thinning plots than on the unmanaged plot. Saw log and pulpwood heights were significantly higher on low thinning than unmanaged plots. Both board-foot and cubic-foot volumes were significantly higher on the thinned than unmanaged plots, 72 and 36 percent, respectively.

DISCUSSION

Thinning has long been recommended as a method of concentrating growth on residual trees. More recently, crop tree management has been advocated as an alternative to area-wide thinning (Perkey and others 1994). However, few studies have compared the long-term impact of various thinning methods on stand structure and growth. Sixty years after a cleaning reduced basal area by 61 percent in an 8-year-old stand in the southern Appalachians, basal area was 23 percent higher and volume 26 percent higher in the cleaned than uncleaned sections (Della-Bianca 1983).

Fifty-five years after an intense surface fire in central Connecticut killed 84 percent of stems and decreased basal area by 38 percent (analogous to a heavy low thinning), stand basal area and density were higher in burned than unburned

Table 6.—*Characteristics of saw log trees (> 10.5 inches dbh) by thinning method at stand age 76 on the Housatonic Blue Ribbon plots, Connecticut*

	Thinning method		
	High	Low	None
Tree grade (stems/acre)			
Grade 1	7.4	6.0	3.5
Grade 2	14.8	16.7	12.9
Grade 3	19.1	22.0	28.2
Ties, timbers, and culls	1.9	4.7	12.9
Upland oaks[1]			
Diameter (inches)	14.3 a[2]	13.8 ab	13.4 b
Sawlog height (feet)	34 a	32 a	30 a
Pulpwood height (feet)	66 a	64 ab	63 b
Board-ft volume (Int. ¼)	132 a	115 ab	105 b
Cubic-foot volume	35 a	32 ab	30 b
Sample size	73	77	52
Non-oak species			
Diameter (inches)	13.2 a	13.0 a	11.5 b
Sawlog height (feet)	34 ab	40 a	26 b
Pulpwood height (feet)	69 ab	73 a	68 b
Board-ft volume (Int. ¼)	109 a	116 a	66 b
Cubic-foot volume	30 a	30 a	22 b
Sample size	51	77	28

[1] *Species are listed in Methods.*

[2] *Values within a row with the same letter were not significantly different at $p \leq 0.05$ (upper case), or $p \leq 0.10$ (lower case).*

sections (Ward and Stephens 1989). Both studies reported that the short-term (≤ 10 years) impact was to reduce stand basal area. Metzger and Schultz (1984) reported differences in understory composition had diminished 50 years after initial thinning method with different harvest methods (clearcutting, group selection, individual tree selection). Long-term research is essential because short and medium-term changes can be ephemeral or even deceptive.

This study found the long-term impact of both high and low thinning on residual stand basal area growth has been minimal (fig. 1). Basal area growth between stand ages 42 and 76 was slightly higher on low thinning plots. A 10-year study in 53-year-old Allegheny hardwood stands found that annual basal area growth was higher on low thinned than on high thinned and unthinned plots (Marquis and Ernst 1991). The long-term impact of thinning on stand volume of oak sawtimber was minor (table 5). There were fewer, but larger, sawtimber oaks on the thinned plots than on the unmanaged plots. In contrast, thinning, especially low thinning, increased the stand volume of non-oak sawtimber.

High and free thinning increase diameter growth rates (Sonderman 1984, Leak and Solomon 1997). Not surprisingly, this study also found that diameter, merchantable heights, and volumes were generally higher on both high and low thinning plots than on the unmanaged plot (table 6), significantly higher for oak sawtimber.

Low thinning is seldom prescribed in eastern hardwood forests. While low thinning had little or no effect on residual tree growth in Pennsylvania (Marquis and Ernst 1991) and Ohio (Dale 1975), a positive growth response has been reported on the western edge of the Central Hardwood Region (Dale 1975). Dale (1975) attributed the differential response to lower precipitation in Missouri and Iowa compared with the more eastern states.

The increased growth of non-oak trees, mostly aspen, to low thinning observed in this study (table 6) is probably not related to increased moisture availability. The increased growth is more likely attributable to the removal of midstory trees, such as red maple. The shading produced by midstory trees accelerates pruning of lower branches, especially in shade intolerant species. Low thinning removed midstory trees allowing residual trees to retain a longer live crown and grow larger boles.

Earlier studies have suggested that the economic benefit of increased diameter growth following release may be offset by reduced clear bole length (Lamson and Smith 1978, Sonderman 1984). In contrast, saw log and pulpwood heights on the Housatonic Blue Ribbon plots were not reduced by early thinning (table 6). Oak sawtimber tree grade was also higher on the thinned plots than the unmanaged plot). This contradiction may be a function of the shorter observation periods (< 10 years) of the aforementioned studies. Trimble (1973) reported that thinning retarded stem pruning of yellow-poplar for several years, but after 7 years thinning had no effect on the length of clear bole.

This study further reinforces the futility of releasing trees in lower canopy positions (Trimble 1974). Using the crown class transition in table 3, to obtain 100 upper canopy crop trees would require releasing 870 intermediate trees compared with only 104 dominant or 131 codominant trees. Crown class regression appears to be a common problem of both released and unmanaged stems for oaks (Lamson and Smith 1978) and other hardwoods (Smith 1979). It is not prudent to select intermediate trees for crop tree release or as the residual trees left after a thinning.

Previous research has suggested that thinning method affects stand composition. Thinning from below has favored an increase in mid-tolerant species (Graney 1988, Ward 1992) and shrubs (Kirkham and Carvell 1980), while thinning from above favors tolerant species (Sander 1979, Abrams and Scott 1989). Low thinning resembles natural thinning in the absence of disturbance—higher mortality in the smaller than larger size classes.

Thinning from below increased shrub density in this study (fig. 2). Thinning from below also reduced density of red maple in the large sapling and small pole-size classes. Not surprisingly, this study found that high thinning increased the number of pole and sawtimber sized red maples. Therefore, commercial thinning, which often resembles high thinning because of the necessity of recovering cost, encourages growth of low value species such as red maple and American beech.

SUMMARY

This study found that both high and low thinning concentrate growth on selected crop trees with an improvement in tree quality. However, there

was no increased stand volume or value that could justify the high cost of thinning/cleaning in sapling stands. Therefore, the choice of thinning method should depend on desired intermediate product(s) and desired future stand composition. Changes in future stand composition induced by different thinning methods will not only affect the quality and variety of future forest products, but will also affect the quality and variety of recreational opportunities and wildlife habitats (Scanlon 1992, Annand and Thompson 1997). A small woodlot owner may prefer a low thinning prescription; cutting large saplings and poles for fuelwood on weekends. Financial considerations will limit many landowners and forest managers to wait until trees in the lower crown classes have grown large enough for a commercial pulpwood operation.

The general results of this research suggest that forest managers should consider the long-term effects of different intermediate cuttings on the composition of the sapling strata. Sapling composition at the time of stand regeneration can have a large influence on the composition of the future stand. Therefore, forest managers should remember that the choice of intermediate cutting prescription can have the long-term consequence of affecting the composition of the next forest.

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LITTERFALL OF TWO SOUTHERN APPALACHIAN COVE HARDWOOD FORESTS

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The current study is a component of a larger study interested in determining ecological change of the mixed-mesophytic forest from the Southern Appalachian Mountains to its far most western branches occurring in Illinois. Although there are numerous reports in the literature on litterfall, no reports were found describing litterfall within undisturbed old growth cove hardwood forest of the Southern Appalachian Mountains. The main purpose of this study was to determine annual litterfall rates for an undisturbed old growth cove hardwood forest and second growth cove hardwood forest of the Southern Appalachian Mountains and to compare the findings for the two forests of different successional stages.

The mixed-mesophytic forest of southern Appalachia contains a floristic diversity unmatched by any other forest type in the eastern United States. Braun (1950) described the southern Appalachian region as the most complex and oldest association of the Eastern Deciduous Forest as well as the area where all other deciduous forests originated. This is further reinforced by Whittaker's (1956) statement that, "The greater ranges of elevation and moisture conditions of the Smokies imply greater floristic diversity, as well as greater variety of vegetation types" than other areas of the southern Appalachians. Thus, depicting the Great Smoky Mountains as the geographic center for the forests of the eastern United States.

Similar studies have been performed in the Southern Appalachian Mountains. Bray and Gorham (1964) gave litterfall rates for an oak (*Quercus*)-hickory (*Carya*)-tulip poplar (*Liriodendron tulipifera*) forest in Tennessee on a north facing slope, south facing slope, and level upland. Reported annual litterfall rates are 4,000 kg/ha for the north facing slope, 5,000 kg/ha for the south facing slope, and 5,400 kg/ha for the level upland. They also gave an annual litterfall rate for an oak-hickory-tulip poplar forest in a valley in North Carolina of 5,300 kg/ha and 2,900 kg/ha for a 78-year-old mixed oak stand in Tennessee.

The criteria for study site selection were based on the anthropogenic disturbance history of the stand and, in the selection of the second growth forest, the time in which that disturbance occurred (Pyle 1985, 1988). For this study, the term old growth follows Pyle's (1988) virgin forest definition of having "the absence of written records concerning human-related disturbances and the absence of mapped land use records." Other attributes of old growth cove hardwood forests included having high species richness and diversity, the majority of canopy trees with a minimum of 150 years of age, the presence of logs at various stages of decomposition, woody debris on the ground, standing snags, canopy gaps, trees of uneven age and size, and pit and mound microtopography (Ellison 2001). Second growth cove hardwood forest characteristics include young, 50 to 100 years old, primarily even-aged stands of tulip poplar due mainly to anthropogenic disturbance such as settlement, farming, or logging (Busing 1993).

Albright's Grove (35º 44′ N; 83º 17′ W) was selected as our old growth cove forest and is located in the northeastern section of the Great Smoky Mountains National Park in east Tennessee (USGS 7.5′ quadrangle: Mt. Guyot, Tennessee and North Carolina). The stand falls between two watershed-draining streams known as Dunn and Indian Camp creeks. Albright's Grove has an average elevation of 975 m and lies on a north-facing slope. The forest composition is characteristic of mesic old growth cove hardwoods, with the dominant canopy species being Eastern hemlock followed by Carolina

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silverbell (see table 1 for scientific and common names). Other important species include American beech, yellow buckeye, Fraser magnolia, yellow birch, white basswood, red maple, and tulip poplar. Some of the largest trees found at the site are tulip poplar.

The only known disturbance to have occurred at Albright's Grove is the infestation of chestnut blight. However, in his 30-year study of stand change, Busing (1993) indicated that given the lack of canopy opening, which in turn did not allow for abundant regeneration of tulip poplar, the disturbance caused by the effects of the blight was not as severe in Albright's Grove as found in other areas of the park.

The Tremont site (35º 42′ N; 83° 41′ W) was selected as our second growth cover forest and is located 30 km west of Albright's Grove. The Tremont site is found on a north-facing slope of Brier Ridge between the Thunderhead Prong of the Little River and Shut-in Creek (USGS 7.5′ quadrangle: Thunderhead Mountain, Tennessee and North Carolina). The average elevation of the Tremont site is 914 m. The Tremont site is unique in that it lies in the area of the Great Smoky Mountain Park where the final harvest was completed in 1939 (Defoe 1994). Being the last area of the park to be logged provides opportunity to compare an old growth cove hardwood forest with the most recently regenerated second growth cove hardwood stand.

The Tremont site's composition is distinctive of a typical second growth cove hardwood forest (Busing 1993). The canopy is dominated by the pioneering tulip poplar, and interspersed with another known pioneer species, black locust. Other species occurring at the site but not as equally important in the canopy are yellow buckeye, yellow birch, sugar maple, red maple, Fraser magnolia, white basswood, Carolina silverbell, black cherry, and Eastern hemlock. Due to the relatively young age of the stand, Eastern hemlock has not become a canopy dominant, allowing more light to penetrate through the canopy and reach the understory vegetation. The increase in light allows for abundant growth of tree seedlings and saplings, shrubs, and herbaceous plants in this second growth site.

Table 1.—*Annual litterfall amounts by tree species and selected components in kg/ha. Species followed an * are significantly different ($p < 0.05$) between the two sites.*

Species		Albright's Grove	Tremont
Acer pennsylvanicum L.	Striped maple*	2.38	10.66
Acer rubrum L.	Red maple	23.70	64.82
Acer saccharum Marsh.	Sugar maple	81.92	68.48
Aesculus octandra Marsh.	Yellow buckeye*	34.24	232.39
Betula alleghaniensis Britton	Yellow birch	50.07	16.82
Fagus grandifolia Ehrh.	American beech*	102.59	0.28
Halesia tetraptera Ellis	Carolina silverbell*	756.85	123.77
	Herbaceous litter*	21.07	293.63
	Insects	9.82	6.16
Leucothoe fontanesiana (Steud.) Sleumer	Dog-hobble	0.06	0.83
Liriodendron tulipifera L.	Tulip poplar*	53.55	1,974.62
Magnolia accuminata L.	Cucumber tree	3.87	4.37
Magnolia fraseri L.	Fraser magnolia*	73.17	1.22
	Misc. petioles*	1.04	98.48
	Misc. plant parts*	74.30	405.63
Prunus serotina Ehrh.	Black cherry*	2.29	36.55
Quercus prinus L.	Chestnut oak*	0	0.29
Quercus rubra L.	Red oak*	0	3.38
	Reproductive litter	331.95	273.34
Rhododendron maximum L.	Rosebay	1.88	0.23
Robinia pseudoacacia L.	Black locust*	0	48.87
Tilia heterophylla Vent.	White basswood	50.89	11.61
Tsuga canadensis	Eastern hemlock*	899.75	29.29
Viburnum spp. L.	Viburnum spp.*	0	11.06
	Woody litter	1,541.23	1,569.35

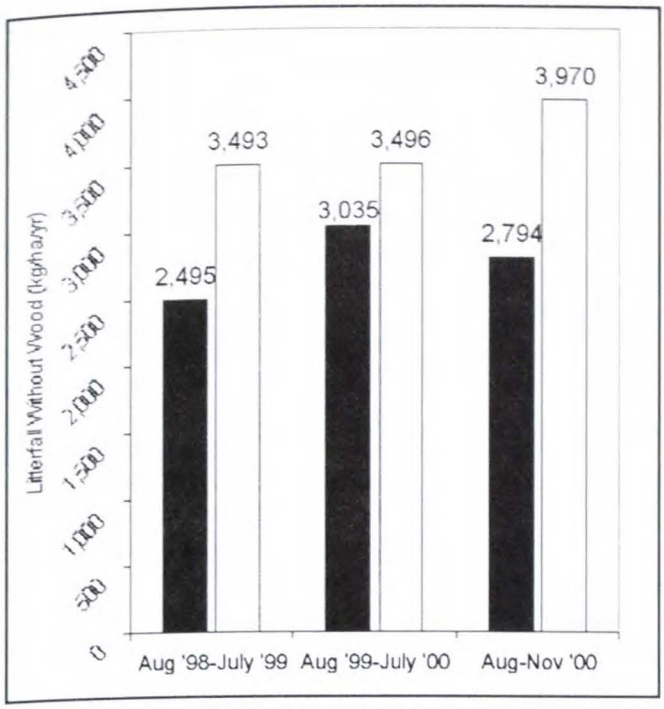

Figure 1.—*Annual litterfall excluding wood for an old growth cove forest (Albright's Grove, closed bars) and a second growth cover forest (Tremont, open bars).*

Aluminum littertraps were assembled and placed in the field at Albright's Grove and Tremont at the end of July 1998. Littertrap dimensions measured 1-m-square by 15-cm-deep. A bottom piece constructed of hardware cloth covered with window screen allowed precipitation and snow melt to drain through the trap. A total of 36 littertraps were placed at each site in 3 rows of 12 on north to south transects. Each littertrap was spaced 15 m apart encompassing a 4,950 m square area.

Monthly forest litterfall was collected from each littertrap beginning in August 1998 and concluding at the end of November 2000 for a total data collection period of 28 months. After each collection the litter was transported to the University of Illinois and dried 85º C for 48 hours. All litter for each trap was weighed to the nearest hundredth of a gram and totaled for the month. Every third bag of litterfall was separated by species and components. Litterfall components were separated by tree and shrub species, herbaceous leaves, woody material > 8 cm in diameter, reproductive material including flowers, seeds, and cones, as well as insects and insect frass. Each component of the litterfall was weighed and totaled for the month. Litterfall amounts were determined with and without woody material because of the variability in woody litterfall that occurred throughout a growing season.

Oven-dry weight data were analyzed using SAS (Statistical Analysis System, Inc., Cary, NC) to perform the two-way analysis of variance. Statistical tests for data normality, including the Shapiro-Wilk test, were administered as well as tests for variability among the data. Tukey's Studentized Range test was applied testing for differences in the amount of litterfall occurring between the two sites, among months, and an interaction of site and month.

An annual litterfall rate was calculated for Albright's Grove and Tremont by summing the oven-dry weight occurring in each month (g/m^2), averaging the amount for the complete 28 months of the study, and converting the total to kg/ha. In the yearly averaging of litterfall, the last 4 months of the study, August to November 2000, were weighted in proportion to the rest of the year based on the corresponding site's amount of litterfall that occurred in the previous 2 years of the study.

For all three litterfall seasons, August 1998 to July 1999, August 1999 to July 2000, and August to November 2000, litterfall without wood was consistently higher at the second growth site (fig. 1). This difference can be attributed to the second growth site's younger stage of succession.

An exceptionally large amount of litterfall occured in the third season August to November 2000, especially at the Tremont site. More litterfall accumulated in this 4-month time period than what was collected during one full year for the previous two seasons. This may be attributable to drought like conditions that occurred earlier in the year. As recorded at the Knoxville National Weather Service station northwest of Tremont, temperatures followed a normal cycle throughout the study period; however, precipitation was noticeably lower during the months of June and July 2000 than during June and July 1999 (fig. 2).

Calculated across the 28-month study, the old growth forest of Albright's Grove had an annual total litterfall rate of 4,808 kg/ha and an annual litterfall rate excluding wood of 3,028 kg/ha (fig. 3). Both the total and litterfall without wood for the second growth forest exceeded the amounts occurring in the old growth forest. The Tremont site had an annual total litterfall rate of 5,751 kg/ha and an annual litterfall rate excluding wood of 4,136 kg/ha.

Significant differences were found in the annual amount of litterfall for some but not all components between Albright's Grove and Tremont

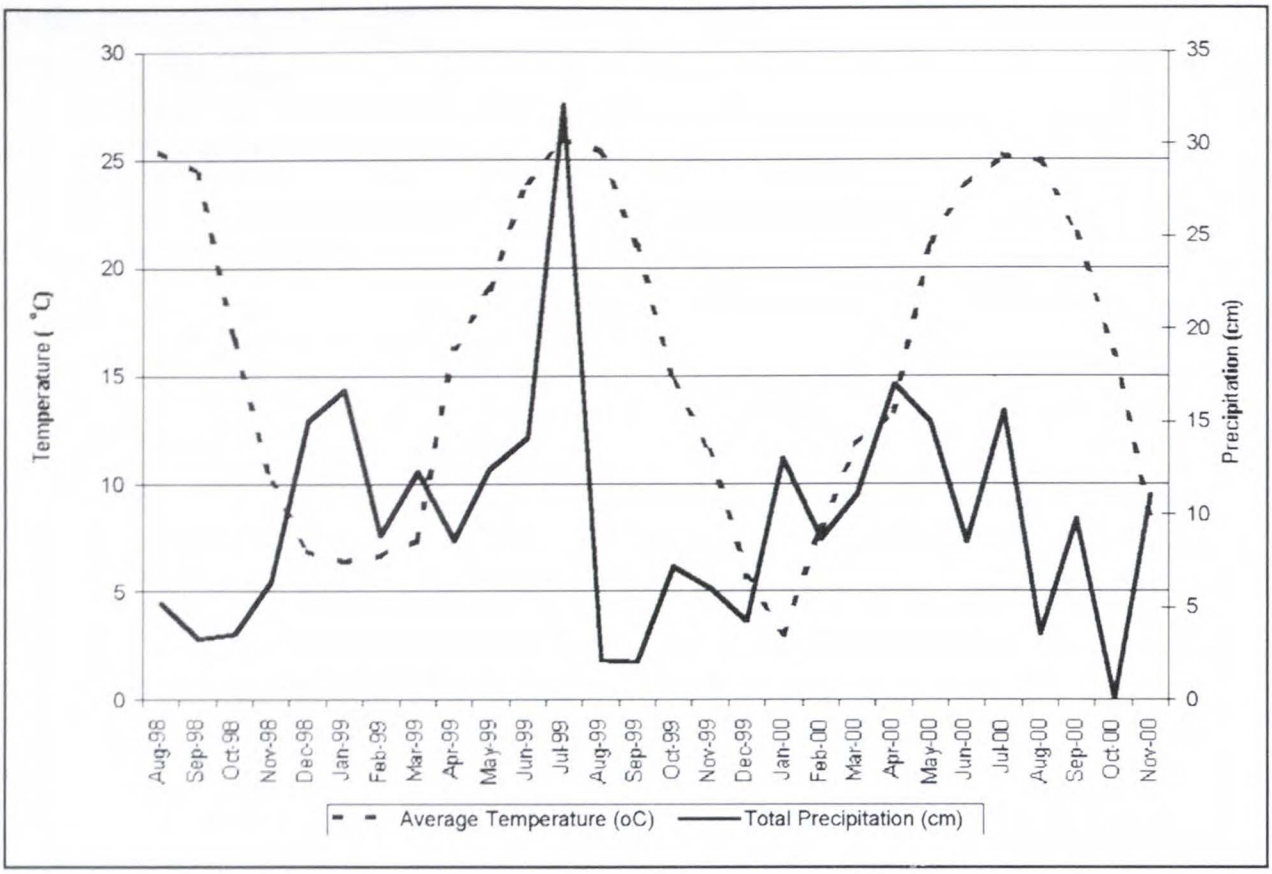

Figure 2.—*Precipitation (dotted line) and temperature (solid line) gathered by the National Weather Service in Knoxville, TN.*

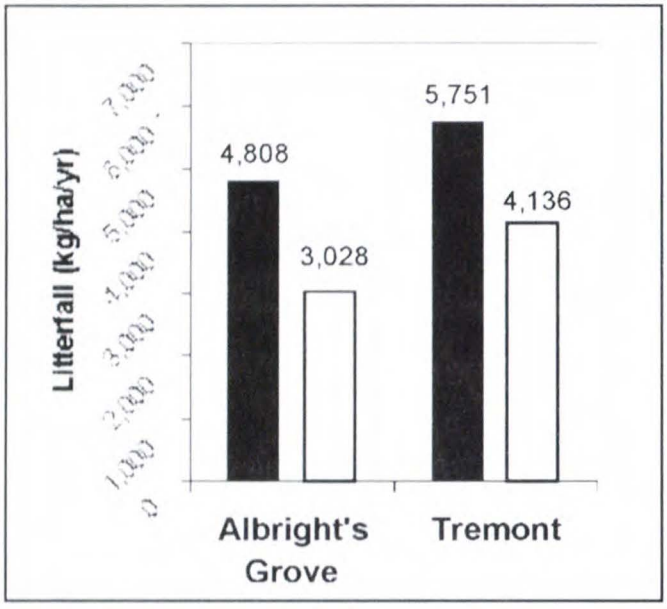

Figure 3.—*Annual litterfall rate with (solid bars) and without wood (open bars) for an old growth cove forest (Albright's Grove) and a second growth forest (Tremont) the Southern Appalachians.*

(table 1). Wood litterfall < 8.0 cm in diameter was not significantly different ($p < 0.05$) between the two sites. Albright's Grove accumulated 1,541 kg/ha/yr while Tremont averaged 1,569 kg/ha/yr. Woody material accounted for 37 percent of the total litterfall at Albright's Grove and 30 percent at Tremont. The greater amount of woody litterfall at Albright's Grove is to be expected due to the forest's advanced stage of succession.

Eastern hemlock litterfall, with an annual rate of 900 kg/ha, was significantly greater at the old growth site than at the second growth site, which had an annual rate of only 29.3 kg/ha (table 1). At Albright's Grove, Eastern hemlock accounted for 22 percent of the total litterfall and 35 percent of the litterfall without wood included. The percentage of Eastern hemlock litter in the litterfall with and without wood at Tremont was found to be 0.5 and 1 percent, respectively. Eastern hemlock is known as a climax species, thus these differences are attributed to the different successional stages of the two forests.

Tulip poplar litterfall was also significantly different between the two sites (table 1). Tulip poplar litterfall accounted for 37 percent of the total litterfall occurring at the second growth Tremont site and only 1 percent at the old growth Albright's Grove. Tulip poplar litterfall at Tremont accounted for 53 percent of litterfall without wood and at Albright's Grove tulip poplar litter accounted for just 2 percent. The difference can be ascribed to tulip poplar being a pioneer species, which would explain its heavy influence at the second growth Tremont forest.

Litterfall rates for Carolina silverbell was also different between Albright's Grove and Tremont (table 1). Litterfall from this species was found to be the third most abundant form of litterfall at Albright's Grove. In the total litterfall occurring at Albright's Grove, Carolina silverbell composed 18 percent, and at Tremont the value was 2 percent. For litterfall without wood at Albright's Grove, Carolina silverbell attained a value of 29 percent and at Tremont there was 3 percent occurring.

The old growth cover forest produced less herbaceous litter than the second growth cove forest (table 1). Herbaceous litter at Albright's Grove, with a rate of 21.1 kg/ha/yr, comprised only 0.5 percent of the total litterfall and nearly 1 percent of the litterfall without wood. Herbaceous litter at Tremont, with a rate of 293.6 kg/ha/yr, comprised 5 percent of the total litterfall and 8 percent of litterfall without wood included. The greater amount of sunlight penetrating the younger canopy of Tremont would allow for the growth of more herbaceous plants than compared to the dense canopy of the undisturbed old growth forest of Albright's Grove.

No differences were found in the annual amount of reproductive litterfall between Albright's Grove and Tremont (table 1). Albright's Grove, with a rate of 332 kg/ha/yr, accumulated enough reproductive material litter to comprise 8 percent of the total litterfall. Tremont's annual accumulation of reproductive material, 273 kg/ha/yr, made up 5 percent of the total litterfall for that site. For litterfall without wood, reproductive material at the old growth site comprised 13 percent, and 7 percent at the second growth site.

In conclusion, the younger second growth forest at Tremont produced more litterfall annually than did the undisturbed old growth forest at Albright's Grove. Compared to the old growth forest, the second growth forest annually produced 943 kg/ha more total litterfall and 1,108 kg/ha more litterfall without wood. As observed, we expect more litterfall produced by the pioneering tulip poplar in the second growth forest. The same is expected of the climax species Eastern hemlock in the old growth Albright's Grove. Interestingly, there was no significant difference in reproductive and woody materials between the two sites. However in this study woody litterfall was limited to the parameters of fine litterfall, or woody material < 8 cm in diameter and the larger branches and fallen trees that occur more often in a climax stage of succession were not included.

The litterfall rates for both study areas, Albright's Grove and Tremont, were of comparable amounts to litterfall rates of similar studies in the area. Albright's Grove, with an annual litterfall rate of 4,808 kg/ha, has a greater litterfall rate than that of a study of a comparable forest type with the same north facing aspect reported by Bray and Gorham (1964). Tremont's annual litterfall rate of 5,751 kg/ha is greater than all of the studies reported by Bray and Gorham (1964) of similar forest types in the Southern Appalachian Mountains.

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CANOPY DISTURBANCE INTERVALS, EARLY GROWTH RATES, AND CANOPY ACCESSION TRENDS OF OAK-DOMINATED OLD-GROWTH FORESTS

James S. Rentch and Ray R. Hicks, Jr.[1]

ABSTRACT.—Using a radial growth averaging technique, changes in growth rates of overstory oaks were used to quantify canopy disturbance events at five old-growth sites. On average, at least one canopy disturbance occurred on these sites every 3 years; larger multiple-tree disturbances occurred every 17 years. Although there was some variation by site and by historical period, there has been little significant change in canopy disturbance rates for the 300-year period examined. A review of tree-ring chronologies yielded three growth strategies. For these oaks,

a) the likelihood of originating in a large opening and achieving overstory status before canopy closure is about the same as the probability of requiring a major release, either from
b) a smaller gap that closed, or
c) from the understory.

Half of the cored trees established individually, and 30 percent attained overstory status individually, members of no identifiable larger cohort. For trees that required a major canopy release, residence time in the understory averaged 89, 54, and 50 years for white oak, red oak, and black oak, respectively. These long-term understory residences averages suggest that the abundance of shade tolerant understory species was considerably less before 1900. These results are consistent with presettlement survey tree-tallies, the absence of shade tolerant species in the oldest cohorts, and fact that virtually no oak canopy recruitment after 1900 occurred after an extended period of below-average growth in the understory.

Concerns for oak regeneration in the eastern United States go back as early as the early 1900s. Clark (1993) notes that Hawley's text, *Practice of Silviculture* (Hawley 1946), described a successful 1902 shelterwood cut that favored oak, and concluded that there must have been a shortage of advance oak in that stand. By the 1950s, forestry literature included accounts that detailed problems with regenerating oak on sites where it had been historically dominant. Difficulties were most pronounced on better sites and regional in scope. The earliest accounts considered second-growth, managed stands (Weitzman and Trimble 1957, Carvell and Tryon 1961), however, a similar trend has been noted in old-growth preserves (McGee 1984, Parker and others 1985).

To solve this problem, a variety of forms of stand manipulation targeting overstory light levels, understory light levels, and inter-species competition have been attempted. More recently, manipulative experiments have been supplemented by ecological studies that seek to understand, and then duplicate, the conditions that fostered oak stands in the first place (Schuler 1998). Key components of this understanding are determinations both of the natural disturbance regime at the time of stand initiation (Oliver and Larson 1996) and the historical range of variability of disturbances over the life of the stand (Aplet 2000).

This study uses dendroecological and community sampling techniques to characterize stand-level dynamics of five oak-dominated, old-growth stands in Ohio, Pennsylvania, and West Virginia. We have two objectives. First, what is the historic rate of canopy disturbance and tree release of the five old-growth stands? Intervals

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between years when at least one canopy disturbance occurred, and intervals between years when larger disturbances occurred were examined over the entire tree-ring chronology of each stand. Disturbance intervals were also compared for several historical periods. The second objective focuses on individual tree dynamics. Early oak growth rates and the presence or absence of a major crown release were used to determine modes of oak establishment and canopy recruitment, and by inference, the understory light environment in which trees originated.

STUDY AREAS AND METHODS

Five oak-dominated old-growth stands were selected for study. These include one stand in Belmont County, OH (Collins Woods), one in Washington County, PA (Wrights Woods), and three in central West Virginia (Watter Smith State Park, Harrison County; Murphy Tract, Ritchie County; and Horners Woods, Lewis County). Two 0.45 ha (60 x 75 m) study plots were established at each site and subdivided into twenty 15 x 15 m subplots, with the exception of Watter Smith State Park, which had a single study plot.

A 100-percent inventory of overstory (dbh ≥ 15 cm) and understory (2.5 cm ≤ dbh < 15 cm) stems was made, and importance values (IV) were calculated for each canopy stratum by averaging relative abundance (stems ha^{-1}), relative basal area (BA ha^{-1}), and relative frequency. Importance values are reported as site averages.

At the minimum, the largest tree in each 15 x 15 m subplot was cored at breast height. An average of 34 trees was cored at each 0.45 ha plot (N_{total} = 314), of which 176 were oaks. Cores were cross-dated by matching unique patterns of narrow and wide rings, and measured under a dissecting microscope to the nearest 0.001 mm in conjunction with J2X® software. Tree-ring dating was validated using the program COFECHA (Grissino-Mayer and others 1997), and the two cores for each tree were then averaged to yield a mean chronology for the tree.

DATA ANALYSIS

Canopy disturbances were identified empirically by an abrupt and sustained increase in tree-ring width, using the decadal averaging technique of Nowacki and Abrams (1997): percent GC = $[(M_2 - M_1)/M_1]*100$ percent, where percent GC = percent growth change, M_1 = preceding 10-year mean radial growth (inclusive of the disturbance year), and M_2 = subsequent 10-year mean (exclusive of the disturbance year) (Nowacki and Abrams 1997).

Three levels of canopy disturbance/crown release were recognized, using the criteria of Lorimer and Frelich (1989) and Schuler and Fajvan (1999):

1) major releases were defined as events when percent GC ≥ 100 percent;
2) moderate releases were classified as those with a percent GC of 50 to 99 percent; and
3) small releases resulted in percent GC of 25 to 49 percent.

Major releases correspond to overhead releases of smaller, overtopped saplings and pole-sized trees in canopy openings. Small and moderate releases apply to gap-associated growth increases caused by increased sidelight to trees already in the overstory.

Canopy Disturbance Intervals

Two types of canopy disturbance intervals were investigated:

1) single-tree event intervals, or the time period between those years when at least one canopy disturbance-crown release occurred in the study site, and
2) multiple-tree event intervals—the time period between years when two or more non-adjacent trees underwent a release.

Years in which at least one disturbance occurred were readily identified. For multiple-tree disturbances, a histograph of all events at each site was compiled, and peaks of the curve identified disturbance years. Because sample size varied with time, we used only trees ≥ 150 years old in the disturbance analysis.

The percent GC technique, as defined, assumes that there is a 1-year lag between the disturbance event and release-response. However, a precise correlation does not always exist between peak percent GC as calculated, and the actual disturbance year (Schuler and Fajvan 1999). Differences in individual tree genetics and physiological condition before the disturbance as well as species-level differences may also affect the precise time of response (Lorimer and Frelich 1989, Schuler and Fajvan 1999). To account for the variability in timing of disturbance and peak percent GC, disturbance events were initially assigned a 3-year window (peak percent GC-year ±1 year), and these data were the input for each site's disturbance histograph. Using a 3-year window for disturbance year

potentially inflates the disturbance frequency by a factor of three; thus a single year was assigned to each disturbance based on the peaks of the curve.

We used the FHX2® (Grissino-Mayer 1997) program to determine the frequency of single-tree and multiple-tree disturbances. Weibull median disturbance intervals were first calculated for the entire tree-ring chronology for each site, and then means of different historical time periods were compared to detect within-site changes over time. Disturbance intervals for three time periods were compared based on a general assessment of human impacts and a variable fire regime (Nowacki and Abrams 1997). The presettlement period is defined as years before 1800. The settlement period (1800-1899) corresponds to the period during which extensive land clearing for agriculture, forest harvesting, and associated slash fire occurred. The modern era (after 1900) is distinguished primarily by fire suppression, the introduction of exotic pests and disease (e.g., chestnut blight and gypsy moth), and a dramatic increase in the white-tailed deer population.

Canopy Accession Trends

Three growth trends were identified based on juvenile growth rates (at dbh), overall growth pattern, and the presence/absence of a major crown release (percent GC ≥ 100 percent).

Gap origin

The absence of a major crown release distinguishes this trend (fig. 1a). Trees with average juvenile growth followed by a slightly increasing trend, or trees with above-average initial growth and a flat or slightly declining trend were considered to be direct overstory recruits in high-light conditions (Lorimer and others 1988, Lorimer and Frelich 1989, Frelich and Graumlich 1994, Nowacki and Abrams 1997).

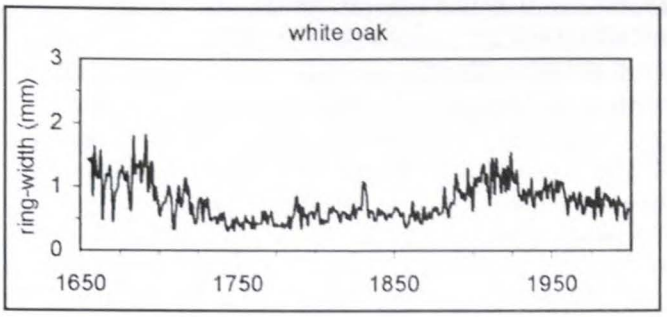
Gap origin-gap release

Radial growth during the first 25 years for these trees was similar to a) above. However, growth was subsequently reduced by canopy closure from above, and then resumed pre-suppression rates after a major release and overstory accession (fig. 1b).

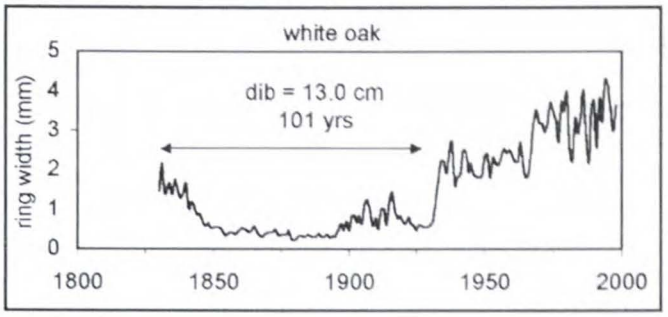
Understory origin-gap release

Initial growth for these trees was at least 50 percent less than the site average (0.5-0.8 mm yr^{-1}), and these trees were considered to have originated in a low light, understory position. Following one, or occasionally, two major releases, growth showed a large increase and no further evidence of overstory suppression (fig. 1c). For trees categorized as gap origin-gap release or understory origin-gap release, dbh (inside bark, dib) at the time of final major release and understory residence time (age at overstory accession minus year of earliest ring) were evaluated by species and by site.

a) Gap origin—no major release.



b) Gap origin—major release.



c) Understory origin—gap release.

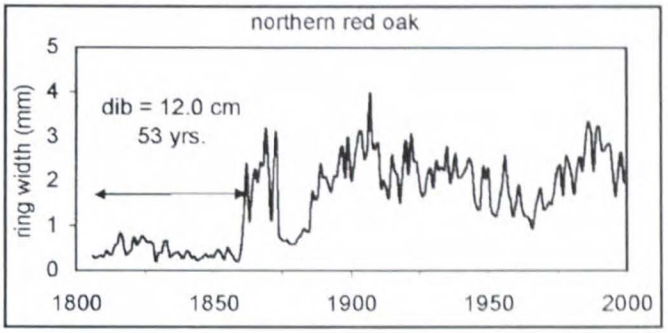


Figure 1.—*Examples of suggested classification of tree-origin and growth trends of canopy oak trees based on early growth rates and presence/absence of a major crown release (percent growth change ≥ 100 percent).*

RESULTS

Table 1 summarizes present stand composition of the five stands. Oaks occupied, at minimum, a codominant position in the overstory stratum. However, oaks were virtually absent from the understory (2.5 cm ≤ dbh < 10 cm), and oak seedlings (stems > 1 m tall, dbh < 2.5 cm)

were present at only one site. Two trends characterized the age structure and oak recruitment at these sites. At Collins Woods and Wrights Woods, establishment of oaks was limited to the period 1650-1875. White oaks (*Quercus alba* L.) were the oldest trees, however the most recent recruitment of this species occurred in 1850; northern red oaks (*Q. rubra* L.) first recruited between 1750-1800 and continued until 1850 at Wrights Woods, and 1875 at Collins Woods.

Both sites showed large ingrowth of shade tolerant tree species in the 1800s, as well as pioneer species such as yellow-poplar (*Liriodendron tulipifera* L.), pignut hickory (*Carya glabra* Mill.), American elm (*Ulmus americana* L), and black cherry (*Prunus serotina* Ehrh.) (table 1). At Watter Smith, Murphy Tract, and Horners Woods, white oak and chestnut oak (*Q. prinus* L.) were the oldest trees, and red oak and black oak (*Q. velutina* Lam.) were recruited beginning in the early 1800s. These sites also showed gradual ingrowth of both shade tolerant and pioneer species in the 1800s, however, in contrast to Collins Woods and Wrights Woods, recruitment of oak species continued during this period.

The oldest trees at Wrights Woods, Watter Smith, and Murphy Tract date to well-defined time periods that suggest these stands initiated in the aftermath of a stand-initiating disturbance. The absence of clearly defined cohorts among the oldest trees at Collins Woods and Horners preclude a reliable estimate of stand origin. Here, the small number of old trees and wider age ranges make it difficult to determine if they are survivors of a large disturbance-initiated cohort, or remnants of a smaller cohort in a stand that has been uneven-aged for centuries. At all sites, shade tolerant species such as sugar maple (*Acer saccharum* Marsh.) and American beech (*Fagus grandifolia* L.) were absent from the oldest cohorts.

Canopy Disturbance Intervals

Table 2 summarizes median disturbance intervals for single- and multiple-tree events for each site. Similarity over time was the most significant finding. Single-tree disturbances occurred about every 3 years, while larger,

Table 1.—*Importance values (IV) for major overstory tree (dbh ≥ 15 cm dbh), and understory tree (2.5 cm ≤ dbh < 15 cm) strata at five oak-dominated old-growth study sites. IV = (relative basal area + relative abundance + relative frequency)/3*

Species	Collins Woods (OH)	Wrights Woods (PA)	Watter Smith (WV)	Murphy Tract (WV)	Horners Woods (WV)
Overstory tree					
red maple	7.4	15.2	13.7	14.9	12.4
sugar maple	34.0	49.0	9.6	0.8	10.7
hickory	2.0	--	8.0	4.0	10.2
Am. beech	21.6	--	12.2	14.4	18.4
yellow-poplar	7.9	--	1.9	--	3.6
black cherry	0.6	10.7	--	--	--
white oak	9.8	15.1	13.7	28.5	10.2
chestnut oak	--	--	4.0	5.0	7.2
northern red oak	7.9	4.8	16.2	6.7	0.6
black oak	--	--	6.9	12.6	16.9
Understory tree					
red maple	3.2	9.2	11.1	30.5	9.3
sugar maple	73.2	79.4	15.2	2.8	37.5
hickory	4.7	1.4	1.9	2.3	2.2
Am. beech	9.2	--	25.0	26.4	31.3
white ash	1.5	--	0.5	0.4	0.9
eastern hophornbeam	6.7	1.2	24.2	0.8	4.0
sourwood			1.0	11.7	2.2
white oak	--	--	--	4.1	--
scarlet oak	--	--	--	0.4	--
chestnut oak	--	--	--	0.6	1.0

multiple-tree disturbances occurred every 17 years. A historical comparison of disturbance intervals revealed a few differences by site. However, even when statistically significant, most differences were small, within the ± 1 year margin of error of the radial growth averaging method, and not ecologically meaningful. There was a slight decline in single-tree disturbance intervals between the presettlement (1700-1800) and settlement (1801-1900) periods, and a slight increase during the modern period (1901-1990, table 2a).

Because the single-tree disturbance frequency is strongly correlated with number of sample trees, increasing sample depth accounts for a portion of these changes. On average, the interval between multiple-tree events slightly increased from an all-site average of 14 years in the presettlement era, to 18 years in the modern period (table 2b). However, the magnitudes of most changes were relatively small, and none were statistically significant.

Canopy Accession Trends

Tree origin and mode of canopy accession were classified as either

a) gap origin-no release,
b) gap origin-gap release, or
c) understory origin-gap release (table 3, see fig. 1 for examples).

For all trees, the likelihood of originating in a large opening and achieving overstory status before canopy closure (N = 85), is about the same as the probability of requiring a major release, either from the understory (N = 66) or from a smaller gap that closed (N = 22).

White oak and red oak best exemplified this trend, with almost the same number of trees requiring at least one release from above (N = 47 and 16, respectively) as those that did not show a major release (46 and 13, respectively). Chestnut oak was more likely to have originated in the shaded understory before release, consistent with its reputation as the slowest grower of these four species (Burns and Honkala 1990). In contrast, a higher proportion

Table 2.—*Median and mean disturbance intervals for five oak-dominated old-growth study sites. Shown are single-tree events and multiple-tree events for the entire chronology, and for three historical periods. Means with different letters are statistically different (F-test for unequal variances).*

Site	Median	-- Means --		
	1700-1990	1700-1800	1801-1900	1901-1990
Single-tree events				
Wrights Woods (PA)	3.0	6.6a	3.1b	3.5ab
Murphy Tract (WV)	2.2	3.1ab	2.3a	4.2b
Horners Woods (WV)	2.8	7.1a	3.0b	2.4b
Watter Smith St. Park (WV)	3.4	5.4a	3.3a	4.6a
Collins Woods (OH)	3.8	6.3a	4.4b	3.8b
Multiple-tree events				
Wrights Woods (PA)	20.7	17.0a	17.3a	18.8a
Murphy Tract (WV)	14.2	12.3a	12.0a	22.3a
Horners Woods (WV)	14.8	17.0a	14.4a	15.8a
Watter Smith St.Park (WV)	15.9	13.2a	18.0a	19.1a
Collins Woods (OH)	14.9	14.0a	13.2a	15.2a

Table 3.—*Summary of growth trends and origin of canopy oak trees based on early growth rates and identification of major release events (GC percent ≥ 100 percent) using radial growth averaging (N = 176). Values represent sums from five study sites.*

Origin-growth trend type	Species			
	white oak	chestnut oak	red oak	black oak
Gap origin-no release	46	3	13	23
Understory origin-gap release	34	9	12	11
Gap origin-gap release	13	2	4	3

of black oaks were successful in using gaps to attain overstory status without being overtopped and then released.

Understory residence time and dib at overstory accession for trees that survived being overtopped are shown in table 4. White oak averaged 89 years before reaching the overstory at an average dib of 15.3 cm. Red oak and black oak attained overstory status sooner (51 and 50 years, respectively) and at smaller diameters (11.8 and 11.0 cm, respectively). Variability of understory residence time was very high. For example, at Murphy Tract, understory residence time for white oak varied from 20 to 215 years.

DISCUSSION

Canopy disturbance rates of the oak-dominated stands of this study are remarkably consistent across a broad geographical area and a 300-year time period. Small scale disturbances occur, on average, every 3 years, while larger events involving more than one tree occur on a 17 year interval. The similarity of canopy disturbance rates was not unexpected. Natural disturbance regimes are strongly correlated with broad geographic patterns of climate, topography, and soils, in the same way that these factors influence the vegetation of a region (Runkle 1990). These landscape-scale factors, in turn, constrain disturbance rates to a fairly narrow and predictable range of values (Runkle 1985).

Canopy Disturbance and Recruitment

The tree-ring record suggests three modes of oak recruitment that correspond to three levels of canopy disturbance:

a) recruitment in large openings after a large stand-initiating disturbance;
b) episodic recruitment in canopy gaps that either remain open long enough for stems to reach the overstory without being overtopped from above, or which undergo a repeat disturbance and second release; and
c) continuous establishment in the understory and overstory recruitment after gap release.

Murphy Tract, Wrights Woods, and Watter Smith State Park originated after large stand-initiating disturbances. Growth trends of these trees were distinguished by the absences of overstory suppression and major crown releases. The absence of shade tolerant tree species in the disturbance-initiated cohort suggests that either some type of disturbance eliminated them (i.e., fire), or they were present but in small numbers.

Oaks were also recruited via smaller canopy disturbances involving two or more trees that occurred, on average, every 17 years. These trees also reached overstory position without being overtopped. However, in contrast to the even-aged structure resulting from stand initiating events, these disturbances resulted in a spatially and temporally dispersed multi-cohort age structure. Large multiple-tree gaps were identified, however two-tree gaps were most common. Oaks were also found that had initiated in the understory under less than free-to-grow conditions. Here early growth rates were consistently less than 1 mm yr^{-1}.

Understory Shade and Recruitment

The length of time these oaks survived in the understory varied by their relative shade tolerance. White oaks, the most tolerant of the four oak species of this study, required on average, 90 years to reach the overstory. Long understory residence time for white oak is consistent with previously published work on this species. Seedlings, saplings, and pole-sized trees may persist for 60 to 90 years under a closed canopy (Minckler 1957, McGee and Bivens 1984), and are therefore in place when canopy gaps occur. Red oak and black oak are generally considered less tolerant of shade than white oak (Burns

Table 4.—*Mean understory residence time (MRT) and dbh (inside bark) at canopy accession for four oak species (N = 89) at five oak-dominated old-growth study sites. Trees included are those classified as gap origin-gap release, and understory origin-gap release.*

	Species			
	white oak	chestnut oak	red oak	black oak
Mean understory residence time (± SE, years)	89±48	38±20	54±24	50±25
DBH (inside bark) at canopy accession (± SE, cm)	13.3±8.3	6.2±5.0	11.7±6.9	10.0±6.8
Number of trees	47	12	16	14

and Honkala 1990), but the percentage of these species that initiated in understory shade (41 and 30 percent, respectively) were comparable to white oak (37 percent, table 3).

The 50-year average understory residence times of red oak and black oak in this study exceed most current estimations of these species' ability to survive heavy shade. Like white oak, seedlings of red oak and black oak readily resprout after dieback, and roots may be 30 years older than stems (Merz and Boyce 1956, Johnson 1994). However, red oak seedlings demonstrate a negative carbon balance under a heavy canopy, and seedlings often die once acorn reserves are depleted (Lorimer 1993). Current experience suggests that survival of aboveground portions of seedlings and sprouts, particularly on better sites, is often very low. In an experiment that removed tall understory vegetation (> 1.5 m), 5-year survival of planted white oak and red oak seedlings was 93 percent compared to only 27 percent survival on control plots (Lorimer and others 1994).

Numerous other studies show that understory light levels are critical for survival and growth of oaks (Carvell and Tryon 1961, Loach 1970), but particularly for red oak, because of this species lower drought and shade tolerance (McGee 1968, Larsen and Johnson 1998). However, while increasing understory light increases oak seedling growth, competing species such as beech, sugar maple, red maple, and black cherry also benefit. The latter two species may, in fact, gain an advantage over oaks due to greater morphological and physiological plasticity, and the ability to respond more rapidly to changing conditions (Gottschalk 1994, Abrams 1998). Therefore, unless a reduction in the level of understory competition accompanies an increase in light to the understory, the temporary advantage available to oaks may be lost.

In the stands of this study, the existence of numerous and variable-sized oak cohorts suggest that they were historically maintained by a unique combination of large and small disturbances, high understory light levels, and low levels of understory competition. However, beginning in the 1800s, conditions changed. Oak replacement largely ceased during the last 150 to 200 years at Wrights Woods and Collins Woods. At Wrights Woods, the youngest white oak and red oak date to 1805 and 1852, respectively. At the remaining three study sites, oak replacement continued through the 1800s and early 1900s, but current understory (seedling-sapling) levels are either low or virtually absent.

The disturbance interval analysis suggests no appreciable change in the creation-rate of canopy gaps. Thus, the primary change must lie with the level of understory light and interspecies competition. There are several lines of evidence to support this hypothesis. Early land surveys attest to forests dominated by oak species (Rentch 2001). Presettlement-era dominance of upland forests in this region by species more shade tolerant than oak was uncommon outside of mesic and protected environments (Russell 1980, Abrams 1992, Lorimer and others 1994, Nowacki and Abrams 1997).

The species composition of the oldest surviving trees does strongly suggest low levels of shade tolerant species in the understory of presettlement forests. Sugar maple and beech are potentially as long-lived as white oak (Burns and Honkala 1990), and seed sources of sugar maple, red maple, and beech were surely present. The canopy disturbance regime described in this study—frequent, small, repeat canopy gaps—should facilitate the presence of shade tolerants in the old-growth cohorts (Runkle 1985, Canham 1990). Nevertheless, none of the study sites had these species in the oldest cohorts. The oldest sugar maples and beech generally date to the 1850s, and as late as 1890 at Wrights Woods. At Collins Woods and Wrights Woods, the earliest establishment of shade tolerant species, 1850 to 1900, was concurrent with the cessation of oak establishment.

The change in understory composition signaled by the recruitment of shade tolerant species in the mid-late 1800s is paralleled by a change in the frequency of oak modes of canopy accession. Figure 2 shows a chronology of oak recruitment by historical period. Before 1900, oaks utilized gap origin-no release and understory origin-gap release strategies in roughly equal proportions. After 1900, 10 of 14 (71 percent) oaks were established in large openings and underwent no overstory suppression. In contrast, only one white oak now in the overstory established in understory shade, and three white oaks originated in a gap and survived being overtopped. These results are influenced by a smaller number of cored trees representing the post-1900 time period, however these trees generally comprise a 100 percent inventory of oaks in 15 cm or greater size-classes.

Evidence from the old-growth oak forests of this study suggests that, for a time, they utilized a relatively constant canopy disturbance rate to replace themselves in kind. Low levels of shade

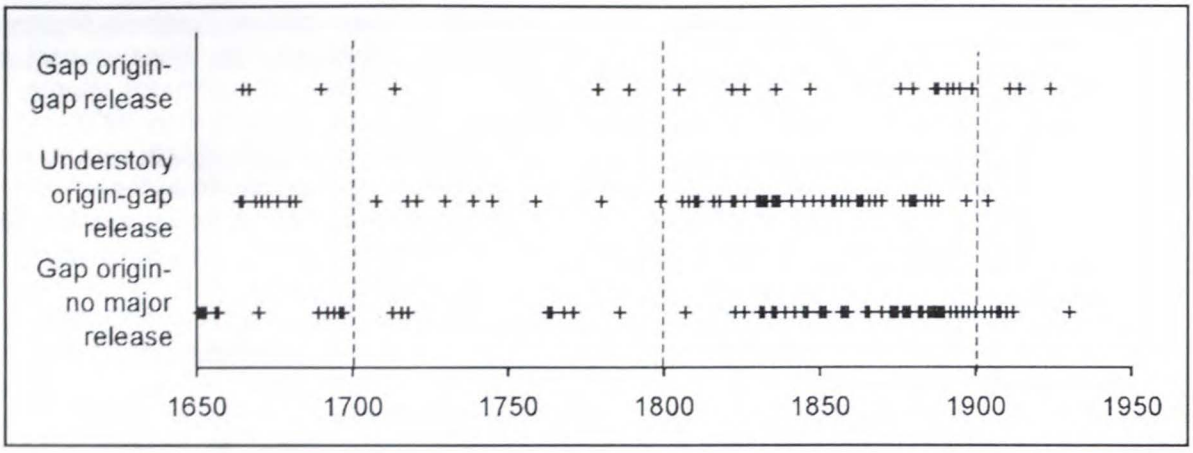


Figure 2.—*Chronology of oak tree-origin by mode of canopy accession for five oak-dominated old growth study sites.*

tolerant species in presettlement land surveys, their absence from the old-growth cohorts, their ingrowth in the mid-1800s, and the concurrent decline of oak establishment as understory species, all suggest substantial changes in the understory species composition over the time period reviewed.

The consequence of understory change was an increase in understory shade, a decline in oak sapling abundance, and a decline in the pool of oak stems that could utilize canopy openings to retain these species share of the canopy. The results of this study support Lorimer's (1993) hypothesis that the principal bottleneck is more a function of the physiological limitations placed on oak advanced regeneration by dense low overstory shade than low levels of oak germination.

CONCLUSION

Because of their antiquity, the old-growth forests of this study provide a valuable source of long-term data on individual tree and stand dynamics of oak forests of this region. These data, in turn, may contribute to a better understanding of the ecological foundations of the oak regeneration problem. Although there was variation by site and by historical period, there have been no significant changes in canopy disturbance rates for the 300-year period reviewed.

Using a variety of growth strategies and canopy opening-sizes, oak components have been maintained in a relatively stable, uneven aged condition. However, current sparseness of sapling-size oak advance regeneration suggests that the future oak composition is in question, particularly on better sites. Oaks' ability to persist in the understory for long periods of time constitutes the key difference between the past and present condition of these stands. Average understory residence times suggest that past understory light levels and the competitive environment allowed oak seedlings and saplings to survive until release. Although there is some historical precedence for these values, few contemporary second-growth forests contain surviving understory oaks of this age, particularly red oak.

These results have implications for the management of oak-dominated old-growth forests, as well as for oak silviculture in general. Most silvicultural prescriptions for oak forests propose even-aged management. The shelterwood system is currently preferred, and this system's manipulation of overstory density, understory light levels, and the level of understory competition is consistent with factors that this study has identified as crucial in the development of these old stands.

The preference for even-aged systems is, in part, due to the even-aged condition of most second-growth forests, and the difficulty of using an uneven-aged selection system without continuous cultural treatments to control more tolerant species. Yet the variety of growth strategies exhibited by these old trees, and the age structures of these stands suggest that continuous "natural" cultural treatments were in fact an integral component of the historic disturbance regimes of these forests, and the key to establishment and survival of a competitive oak understory, and eventual accession to the overstory.

Changes in the historic fire regime best account for the changes in stand composition and structure suggested by this study. No other form of

surface-level disturbance discriminates in favor of oak species and against maple, beech, and other typical understory competitors. Most of the long-term fire chronologies are for areas at the periphery of the Central Hardwood region. These studies suggest a relatively frequent recurrence of low intensity surface fires during presettlement and settlement times; the few studies of oak dominated forests in the study area confirm this. Additional fire chronologies of older forests are a needed area of research. In addition, stem reconstructions of trees that initiated in understory shade would be useful in verifying height growth patterns proposed by this study.

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FROM SAVANNA TO CAMPUS WOODLOT: THE HISTORICAL ECOLOGY OF FARM WOODLOTS IN SOUTHERN ILLINOIS

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ABSTRACT.—The historical ecology of Thompson Woods, a 4.1 ha forest remnant on the campus of Southern Illinois University-Carbondale, was investigated through stand structure analysis, dendroecology, and historical records. Historical records indicate the area was a savanna ecosystem prior to European settlement dominated by large, open grown mixed oak-hickory trees. No trees in the current stand predate 1850 suggesting that most of the trees standing at settlement were harvested for the stave industry. Following this cutting, shagbark hickory and white, post, black, southern red, and black oaks were recruited on the site and formed the current 140-year-old overstory. During the latter part of the 18th century, the stand area was apparently managed for firewood and an occasional saw log along with fires and intermittent grazing to maintain the open characteristic of the understory. In the early 20th century, the area was isolated from adjacent forested areas and removal of damaged overstory individuals coupled with fire exclusion activities has fostered the increase of fire intolerant, more mesophytic species. Currently, restorative management activities are focused on removing undesirable midstory sugar maple and beech while conducting prescribed burns in the understory to foster oak-hickory recruitment.

Oak savanna was once a major vegetation type across large portions of the Midwest stretching from southern Wisconsin to western Tennessee (Cottam 1949, Bray 1960, Nuzzo 1986, Heikens and Robertson 1995). In general, savanna is a transitional community between areas of forest and grassland (Nuzzo 1986). Cottam (1949) characterized savanna as a "community consisting of an overstory of widely scattered trees with a ground cover consisting largely of prairie grasses." In southern Illinois, these open communities were historically classified as dry-mesic savanna on fine-textured soils of the Till plains; dry-mesic sand savanna; or dry and mesic upland sites of the Ozark Hills, Shawnee Hills, and Cretaceous Hills (Anderson and Anderson 1975, Fralish 1976, Nuzzo 1986, Heikens and Robertson 1995).

Ecologists have generally attributed recurring fire and edaphic factors to the long-term maintenance of these unique ecosystems (Cottam 1949, Nuzzo 1986). Native American use of fire, coupled with lightning caused fires, is widely accepted as a contributing factor in the prehistoric development of savanna in southern Illinois (Anderson 1983). However, fire scar evidence in this region is scarce at best and few researchers have successfully constructed fire histories (Robertson and Heikens 1994).

Site characteristics, such as aspect and shallow soil, also have been suggested as factors responsible for savanna development (Anderson and Anderson 1975, Fralish and others 1991). Before settlement, savanna represented up to 18 percent of the forested landscape across southern Illinois (Anderson and Anderson 1975). However, following European settlement, cessation of fire, widespread logging, and continual grazing of the forest contributed to the near loss of savanna forests (Nuzzo 1986). Nuzzo (1986) reported that less than 0.02 percent of the original savanna vegetation existed in 1985.

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Recent land-use history studies have documented past land-use history's role in altering pre-settlement vegetation ultimately shaping modern ecosystems (Foster 1992, Orwig and Abrams 1994, Ruffner and Abrams 1998, Ruffner and Arabas 2000). These studies help identify direct environmental consequences of Euro-American land-uses across the eastern deciduous biome. Within southern Illinois, few studies have investigated the role of land-use history in shaping current forest systems (Fralish 1997). This study was designed to investigate the historical ecology of Thompson Woods, a remnant oak savanna on the campus of Southern Illinois University in Carbondale, Illinois.

Numerous reports of the 1950s-1970s discussed the open nature of this stand and its aesthetic qualities amenable to picnicking and walking for local townspeople. During the late 20th century a series of overstory disturbances removed many of the large oak trees thus beginning a serious decline in the health of the stand (Roth, personal communication). Species composition data collected in the mid-1980s indicated that the oak stand was being replaced by mixed mesophytic species and the historic savanna structure was no longer intact (Roth, unpublished data). Thus, during a graduate Historical Ecology field course in the fall of 2000, students representing several disciplines decided to document the historical ecology of the stand to better understand the historical forces responsible for maintaining the oak savanna and which modern day forces were contributing to its replacement.

STUDY AREA

Thompson Woods is located on the campus of Southern Illinois University at Carbondale in Jackson County, IL. The area has primarily Hosmer series soils characterized as moderately well-drained soils that formed in loess on upland sites (Herman 1979). The site itself is gently sloping with an intermittent tributary of Crab Orchard creek draining the woods. Historically, Thompson Woods was considered an upland oak savanna on the border between the Till plains to the north and the Shawnee Hills to the south (Anderson and Anderson 1975, Edgin and Ebinger 1997).

The area is a 4.1-hectare mixed oak-hickory woodlot composed primarily of decadent southern red oak, black oak, post oak, and shagbark hickory in the overstory (see table 1 for scientific and common names). In contrast mixed mesophytic species such as black cherry, sugar maple, American beech, and a mix of exotic vines and shrubs dominate the understory. The stand is located in the center of campus and is considered by some to be a key part of the Southern Illinois University landscape.

Over time, Thompson Woods has experienced many land-uses, but today is largely dissected by paved pathways and is used for recreation and education. Because of its importance to the University community, Thompson Woods is an ideal area for historical analysis to elucidate the effects of differing land-uses on savanna vegetation over time and to attempt an understanding of the forces needed to begin restoration of this beloved campus woodlot.

METHODS

We compiled the history of human activity on the site from a variety of sources. Pre-settlement Native American occupation was gleaned from local archaeological sources. While no major river system lies near the area reducing the likelihood of a large village, local seasonal camps existed along nearby creeks and numerous rockshelters have been utilized since Archaic periods (B. Butler, personal communication).

Post-European settlement history was investigated using local histories, archival resources, and unpublished memoirs of local businessmen (Boggess 1906, Lentz 1955, Allen 1963). To document land transfers and bills of sale, we searched deeds and tax records archived in the Jackson County Courthouse, Murphysboro, Illinois. In addition, old maps and aerial photographs of the campus provided visual evidence for vegetation change on the study site.

During October 2000, the class collected stand structure analysis data in ten 0.02 ha overstory sample plots placed every 20 m along transects through the woodlot. Overstory data included diameter at breast height (dbh = 1.37 m) by species for all trees (dbh > 8 cm). For each tree species, we calculated a relative importance value (IV) by summing the relative density (density = # tree ha^{-1}) and relative dominance (basal area = m^2 ha^{-1}) and dividing by two (Cottam and Curtis 1956). These importance values refer to the relative contribution of a species to the entire community (Barbour and others 1987). In addition, 44 overstory trees were cored at dbh for age determination and radial growth analysis. Nested circular plots of 20 m^2 and 5 m^2 were used to collect sapling (stems < 8 cm dbh, > 1.4 m tall) and seedling (0.2 m tall < stems < 1.4 m tall) densities within each of the overstory plots.

Individual cores (n = 44) were taken to the laboratory, dried, mounted, and sanded following methods of Phipps (1985). Cores were cross-dated using signature years and verified using the COFECHA program and ITRDLIB (Stokes and Smiley 1968). We used cores from the eight oldest southern red oak trees to construct a mean growth chronology of the site spanning 140 years. We measured annual growth increments with a Velmex measuring system and recorded the measurements with the PJK5 microcomputer program. Each time series was standardized to remove the age-related growth trend by dividing the yearly measured growth values by expected values obtained from fitting a linear regression to measured vales or the mean growth increment (Fritts and Swetnam 1989). These standardized ring-width indices were then averaged to develop the mean stand Chronology (Fritts and Swetnam1989).

RESULTS AND DISCUSSION

The era of European settlement can be divided into two phases, the prep-railroad era and the post-railroad erra.

Pre-Railroad Era (Early 1800s to 1850)

Direct historical data for the pre-railroad era is scanty. At the time of township surveys in 1806 through 1808, inhabitants in the vicinity of Carbondale were not noted. The war of 1812 brought the first wave of settlers to Illinois territory migrating from the east, and especially the south. According to Edmund Newsome in 1894, settlement in Jackson County in 1878 clustered around the Big Muddy River, the only major waterway in the county (Jackson County Historical Society 1997). In 1816, Jackson County was formed in Illinois Territory with a population of 565 inhabitants. According to S. Augustus Mitchell in his 1837 article titled *A Sketch Descriptive of the Situation*, settlement had rapidly continued and by 1835 the population had grown to 2,783 inhabitants (Gates 1979). In a promotional/informational pamphlet entitled *Illinois* in 1837, Jackson County was described as being, "mostly timbered, although it contains many prairies" (Gates 1979).

The Phelps family was the first to settle in Carbondale Township. They cultivated two or three acres of land in section 34 [south of the study area] and subsisted on wild foods gathered from the surrounding woods. As in the county at large, the first settlements were situated along small drainages. Families cleared enough land for subsistence and often, residence was temporary as people moved if crop yields were poor (Wright 1977). As reported by Marten in 1878, less than 10 acres were under cultivation in the Carbondale Township in 1831 (Jackson County Historical Society 1983). Mrs. John Alexander Floyd recalled that when the area was first settled, "there was more trees and hazel thickets than houses" (Wright 1977).

As settlement in the Carbondale Township increased, one of the major timber resources extracted from the forest was fence rails. John Allen (1963) estimated that one third of southern Illinois was encompassed by rail fences. He described the area surrounding the present day University campus as criss-crossed by rail fences. B.G. Root writing in about 1850, described some of the issues with fencing.

> "Fencing is the hardest work which a new settler has to perform. Good white oak rails, laid up in fence, where it is required, are worth from $2 to $3 per hundred. To lessen the cost of fencing, it is very desirable for several friends to settle together, so that the land at first may be enclosed in one common field. 4,704 rails will fence 40 acres: 13,440 rails will fence 160 acres; 28,880 rails will fence one section, or 640 acres" (Gates 1979).

The intensity of land-use during the early settlement period in the area of Carbondale was probably greater than during Native settlement periods. The rising population of permanent farmers whose agrarian practices included fencing their fields began to have an effect on forest structure.

Railroad Era (1850 through 1939)

The town of Carbondale was founded with the sole intent of benefiting from opportunities for commerce afforded by its proximity to the Illinois Central Railroad. On 1850 September 20, the U.S. Congress granted the State of Illinois 2,595,000 acres of public lands for a railroad throughout the State (Gates 1979). This sparked the beginning of intensive use of the forests in Southern Illinois.

Daniel Brush was the driving force behind establishing a town on the railroad at a midpoint between two towns already designated as railroad stations, Makanda and De Soto. By late July 1852 the Illinois Central Railroad had established the rail line and surveyed to the Big Muddy River, in the northeast portion of the county. Daniel Brush sought an ideal town site

and described the vicinity of Carbondale as a "wilderness of forest and dense undergrowth of hazel bushes, wild grape, and running rose vines all over the surface of the ground" (Quaife 1944). However, in a landscape of hilly terrain, the site he chose was level and the railroad was persuaded that the proposed town of Carbondale would be an ideal place for a rail station.

Seizing the opportunity, in the fall of 1853, Brush dismantled a mill he owned near Thebes where the timber supply was already exhausted. He brought all the mill fixtures to Carbondale where he set up a lumber mill with two circular saws and a gristmill near the present day University central quad (Quaife 1944).

Brush described the mill and business as follows:

> "I employed a good many men myself in the building of my mill and storehouse, the depot and a large woodshed for the Company [Illinois Central Railroad], and in preparations for my dwelling house, having the stone foundation laid, the cellar dug out and walled, cisterns excavated, etc.; and having the station to attend to besides, kept me busy. I kept the sawmill doing its best turning out lumber for the buildings I was erecting. I reserved the best for my dwelling house and had it nicely stacked to season and be ready when I should start the work. I had a supply of choice poplar and oak timber on my own land near the mill, while I gathered it in and cut it up as fast as my steam-propelled circular saws could do it. I purchased off farmers and others a large quantity of wood by the cord delivered at the shed for use as fuel in their engines, no coal being then burned by them. It was well for those who were clearing up land nearby as they could turn their timber into money at a paying rate" (Quaife 1944).

Brush's mill was located in close proximity to Thompson Woods (less than 200 m) indicated as "choice poplar and oak timber" in the above list of landholdings. This "supply of ... timber on my own land near the mill" suggests that the woods were standing at the time of writing. This establishment of sawmills for lumber extraction was a common theme across the region. In a letter dated 1855, B.G. Root advertised for the Illinois Central Railroad on the desirability of settling in southern Illinois.

> "Numerous saw-mills are being erected in the timber along the railroad, south of the Big Muddy River. Some are completed, and lumber yards are established at almost every station, where the pine of the North meets the poplar, cypress, black walnut, sycamore, maple and oak, from the South. There are saw-mills in the smaller portions of timber which occur at short intervals in this part of the State, but they are fully occupied in supplying the demand in their vicinity" (Gates 1979).

On July 4, 1854, the railroad track reached Carbondale from the south. The construction of the railroad north of Carbondale, an area where trees were sparse, required imported lumber and timber supplies.

Sutton (1981) reported that "crews of lumber jacks" were sent into the forests of Michigan, Wisconsin, and southern Illinois to supply the demand for cross-ties, bridge timbers, and finished lumber for buildings. Lumber was the second most important commodity, after agricultural products, shipped by the railroad. In many of the prairie regions, north of the study area, lumber for building, firewood for domestic use, and rail for fencing was in short supply. Lumbering in the southern parts of Illinois was especially encouraged to meet demand. The lumber industry developed rapidly, benefiting both the Illinois Central Railroad and those engaged in the lumber business.

In 1857 Carbondale reportedly had two steam sawmills and steam flouring mill and three brickyards (Wright 1977). In 1861, the Civil War broke out and population declined. The Illinois Central Railroad was instrumental during the war in shipping provisions and troops to the southern battlefields. The use of wood fuel in locomotives continued through the Civil War and until about 1895 when most locomotives had converted to coal (Sutton 1981).

The construction of the town buildings was also a draw on forest resources. Not only were the buildings made of lumber, but also sidewalks and streets were constructed of wood planks. In an editorial in the *New Era*, 1867 May 9, John H. Barton complained about the lack of repair of some city streets and that the situation was perilous for pedestrians. Other streets, he noted, "are kept in prime order and not a board nor a nail can be found out of place for several blocks," while others "can only afford a few narrow, rotten boards, perhaps fastened at one end, which renders the way both more unpleasant and dangerous than the bare earth" (Quoted in Wright 1977).

By the last quarter of the 19th century, there were indications that the land had been significantly impacted by European settlement practices. With the settling of the County, large tracts of timberland had been clear felled and game had largely been killed off and forced elsewhere in search of more sparsely populated districts.

Another indication of the degree of impact on forest resources is in the account of a murder. On 1878 June 29, Frank Chapman, a Carbondale resident who owned an interest in a large lumber mill near Vienna [south of the study area] was shot and killed while asleep in his hotel room. While the murderer was never brought to justice, one explanation asserted "operations at the mill had been declining as the timber supply was more and more depleted, and labor troubles were possibly at the bottom of the murder" (Wright 1977). Further in 1880, the Sanders' (Brush) mill was destroyed by fire and never rebuilt (Wright 1977). While the reasons for this are unclear, there were two other mills operating in town, and it may no longer have been profitable to operate a third.

In 1870, Carbondale was chosen for the location of a second teacher's training institution in Illinois. Mrs. Sanders, widow of Henry Sanders, gave 21 acres of farmland for the school. Four years later, in July 1874, the Southern Illinois Normal University was completed and dedicated. Lentz state that

> "[t]his beautiful building, a model of school architecture for the time, which had been erected on the site of Mrs. Sanders' strawberry patch, had, however, a most unattractive setting. Earth from the slight excavation for the first story stone basement had yet to be leveled; landscaping, tree planting, fencing, and pavement had to wait development during the months ahead. The treeless field required grading and fencing; walks must be provided and streets leading to the residential area opened up" (Lentz 1955).

In 1886, Theodore Thompson purchased land adjacent to the university, on the west half of Section 28 and northeast quarter of Section 29. The property included Thompson Woods and the area surrounding Campus Lake. It was called Thompson Lake at that time. The Thompson's were fruit growers, which suggests that the land adjoining the university was largely agricultural. In 1903 Theodore Thompson died, leaving the NW quarter of Section 28 to his wife Lavinia except for 8 acres that were reserved for homes for his four children (Wright 1977).

In fall 1902, the Ayer and Lord Tie Company began operations in Carbondale. Mr. Lord, the proprietor, "told the local people that the plant in Carbondale, when in operation, would have a capacity 25 percent larger than any other plant of similar function in the world" (Wright 1977). This suggests that some regrowth of timber had ensued since the vast cuttings of the 1850s and 1860s. In 1914, white oak from the woods was harvested for use in the stave industry (Karlovich and Niebruegge 1997).

University Campus

In 1940, Lavinia Rendleman Thompson sold her property to the Southern Illinois Normal University. The purchase contract included a restrictive clause stating that Thompson Woods was to be retained in a natural state as a "priceless feature of landscape architecture." As Thompson Woods was incorporated into the University campus it slowly became regarded as a "wilderness area." Despite this perception, over the past 60 years different management activities have been implemented. Various hardwood seedlings, predominately oak, ash, and tulip poplar have been planted. In addition, the University's physical plant has performed various other management operations. For example, in 1980, in the eastern and southern portions of the woods, underbrush was cleared and all trees less than 6 inches in diameter were felled. This resulted in the removal of half of the seedlings planted during the 1970s.

More recently, management activities have included the removal of some non-native tree species and herbiciding of vines and more competitive herbaceous species (Karlovich and Niebruegge 1997). Natural events have played a role in shaping the stand structure. In the summer of 1980, two large windstorms damaged many of the larger trees, especially the oaks and hickories. It is estimated that 30 percent of the woods' total basal area was downed during the two storms (Karlovich and Niebruegge 1997).

Current Vegetation

The Thompson Woods stand is currently dominated by black cherry and elm with minor contributions of black oak, hickory, and American beech (table 1). Relative importance values indicate that black cherry, the largest contributor to the mid-canopy classes, contributes both high basal area and density to the

stand. In contrast, the canopy dominants such as southern red oak and black oak contribute relatively low basal area and densities to the stand.

The canopy of the stand reflects fairly equitable numbers of species across canopy classes (fig. 1). However, these classes also reflect a considerable increase in the number of mixed mesophytic species recruiting into the upper canopy at the expense of the two dominant oak species, southern red and black oak. Indeed, very few oak trees occupy the mid-level canopy positions.

The current diameter distribution also suggests that most oaks exist in the larger diameter classes with only a few red oaks in the 5 to 9.9 cm range (fig. 2). These individuals reflect balled stock plantings by the University to augment poor oak regeneration in the stand. The inverse-J shaped diameter distribution reflects a typical uneven-aged stand comprised of older oak-hickories in the larger diameter classes with transitional mixed mesophytic species in the low diameter classes (Smith 1986).

Stand Age Structure

The dominant overstory trees were all oak species recruited following the cutting of Thompson Woods in the 1850s (fig. 3a). The age structure suggests that the original oak stand probably regenerated via advanced regeneration and/or stump sprouting following the cutting. Initial recruits included the less shade tolerant post and southern red oak, followed 30 years later by the mid-successional white oak (Abrams and others 1995, Shumway and others 2001). Indeed, the stand appears to have developed a

Table 1.—*Summary of current vegetation including basal area (BA) and density (#/ha). Importance values (IV) are the average of the relative dominance and density values*

Species	BA per ha	Rel. dominance	# per ha	Rel. density	IV
Prunus serotina Ehrh. (black cherry)	5.9	26.2	177.1	32.7	29.5
Ulmus L. spp. (elm)	1.9	8.3	73.5	13.6	10.9
Quercus velutina Lam. (black oak)	3.0	13.2	13.4	2.5	7.9
Carya L. spp. (hickories)	2.0	8.7	33.4	6.2	7.5
Fagus grandifolia Ehrh. (American beech)	2.0	9.0	20.0	3.7	6.4
Fraxinus americana L. (white ash)	1.0	4.3	30.1	5.6	4.9
Quercus falcate Michx. (southern red oak)	1.4	6.2	16.7	3.1	4.6
Acer saccharum Marsh. (sugar maple)	1.0	4.4	13.4	2.5	3.4
Liriodendron tulipifera L. (tulip poplar)	0.7	3.0	20.0	3.7	3.4
Sassafras albidum (Nutt.) Nees (sassafras)	0.4	1.9	20.0	3.7	2.8
Gymnocladus dioicus (L.) K. Koch (Kentucky coffeetree)	0.9	4.1	6.7	1.2	2.6
Cercis canadensis L. (eastern redbud)	0.2	1.1	20.0	3.7	2.4
Morus alba L. (white mulberry)	0.3	1.2	13.4	2.5	1.8
Quercus palustris Muenchh. (pin oak)	0.5	2.3	6.7	1.2	1.8
Acer negundo L. (boxelder)	0.1	0.5	13.4	2.5	1.5
Diospiros viriginiana L. (persimmon)	0.2	0.9	10.0	1.9	1.4
Morus rubra L. (red mulberry)	0.2	0.8	10.0	1.9	1.3
Platanus occidentalus L. (American sycamore)	0.4	1.7	3.3	0.6	1.2
Prunus avium (L.) L. (mazzard or bird cherry)	0.0	0.2	10.0	1.9	1.0
Maclura pomifera (Raf.) Schneid. (osage-orange)	0.0	0.2	6.7	1.2	0.7
Celtis occidentalis L. (hackberry)	0.2	0.7	3.3	0.6	0.7
Quercus alba L. (white oak)	0.0	0.1	6.7	1.2	0.7
Liquidambar styraciflua L. (sweetgum)	0.1	0.5	3.3	0.6	0.6
Juniperus virginiana L. (Eastern redcedar)	0.1	0.4	3.3	0.6	0.5
Catalpa Scop. spp. (catalpa)	0.0	0.1	3.3	0.6	0.3
Cornus florida L. (flowering dogwood)	0.0	0.0	3.3	0.6	0.3
Totals	22.6	100.0	541.3	100.0	100.0

savanna type structure following the initial cutting of the 1850's, which was maintained throughout the early 1900s by burning and grazing (Ruffner and Arabas 2000).

Radial analysis coupled with the age-diameter relationships allows us to reconstruct the structure of the stand in the late 1890s (figs. 3a and 3b). Early growth of the post oak indicates high growth rates indicative of a high light environment such as would be found in a recently cut forest. Moderate fluctuations of the ring-with index indicate another canopy disturbance in the mid-1870s, perhaps selective logging or understory burning. Following this increased growth is a regeneration pulse of southern red, white, and post oak that continues into the early 1900s. Following this is a distinct stem-exclusion stage probably fostered by extensive grazing in the forest during the farm establishment period discussed above (Abrams and others 1997, Ruffner and Arabas 2000).

No major recruitment or growth releases occurred through the early 1900s until after 1950 when a significant cohort of mixed mesophytic species were established, timed with a moderate overstory disturbance (fig. 3). No significant oak regeneration has occurred in the last 100 years of the stand's history and it appears to be transitioning towards a later successional stand comprised of black cherry, sugar maple, and beech. Similar predictions have been made for other midwestern and northeastern mixed oak-hickory forests (Lorimer 1985, Abrams 1992, Orwig and Abrams 1994, Ruffner and Abrams 1998, Shumway and others 2001).

The original savanna vegetation of Thompson Woods was essentially lost after the transfer of the parcel to the University. Cause for this shift in vegetation types is equivocal at this time but several variables may explain this. First, the "protection clause" stating that Thompson Woods was to be retained in a natural state as a "priceless feature of landscape architecture" was interpreted quite literally by the University to mean that no "active" management could be implemented to maintain the unique oak savanna structure (P. Robertson, personal communication). Second, the political and ecological mindset of the mid 1950s held fire as an enemy of the forest and thus no University groundskeeper would understand or perhaps accept the inherent need for fire in this ecosystem.

The University, however, understood its limitations to this hand's off approach to

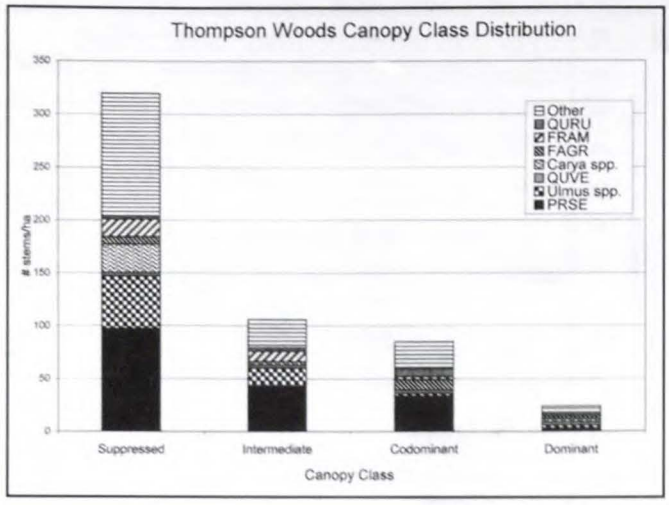

Figure 1.—*Canopy class distribution for species in Thompson Woods, Southern Illinois University campus woodlot. Abbreviations for species are QURU,* Quercus rubra*; FRAM,* Fraxinus americana*; FAGR,* Fagus gradifolia; *QUVE,* Quercus velutina*; and PRSE,* Prunus serotina.

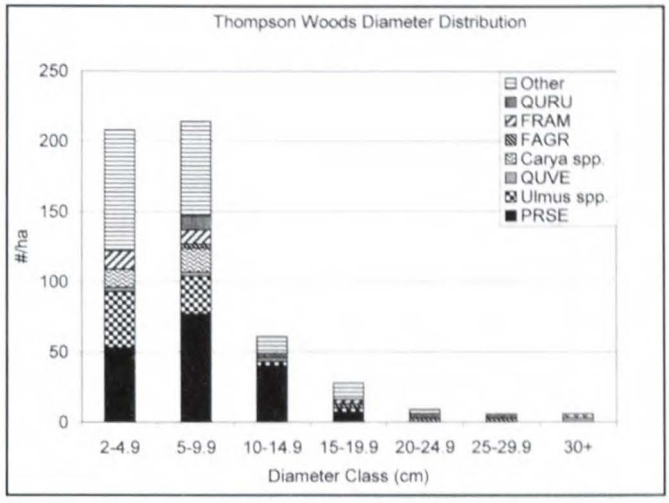

Figure 2.—*Diameter distribution for species sampled in Thompson Woods. See figure 1 for species abbreviations.*

Thompson Woods. Afterall, several times during the 1970-1990s, numerous storms battered the overstory trees and many of these individuals were felled for safety reasons. By the time of these selective fellings, oak regeneration was outcompeted by the more numerous mixed mesophytic species and these disturbances acted only to accelerate the succession of the stand (Abrams and Scott 1989, Abrams and Nowacki 1992).

Future management options include sustained planting of native oak-hickory seedlings that heretofore has met with little success. The

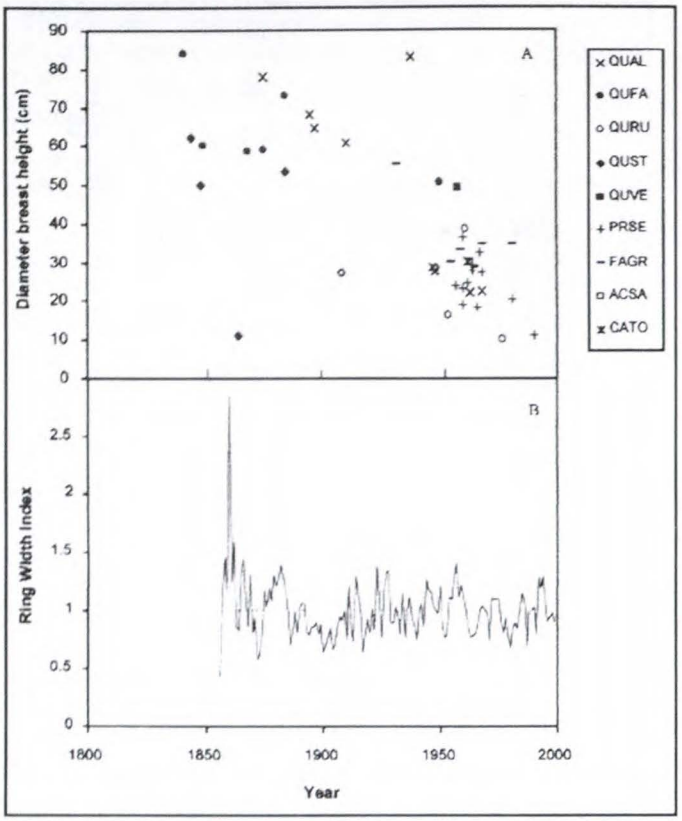

Figure 3.—*Age-diameter relationships for trees cored in Thompson Woods survey. Ring width index for five oldest southern red oak trees.*

primary reason for the failure of these plantings appears to be the lack of adequate sunlight reaching the forest floor. With extremely shaded understory conditions, planted oak seedlings will continue to fail until large gaps are formed in the canopy (Larsen and others 1997). During the summer of 2001, numerous large canopy gaps were formed via felling of overstory sugar maple and beech trees.

While canopy openings may enhance oak survival, successful recruitment into these gaps requires the reduction of competing vegetation, i.e., advanced reproduction of sugar maple, beech, and black cherry. To accomplish this reduction of competing vegetation, a prescribed burn was conducted in October 2001 and another is scheduled for Fall 2002. Understory burns have been shown to reduce small stems of fire intolerant species such as maples, beech, and ash (Brose and Van Lear 1998).

CONCLUSIONS

An oak savanna woodland developed as a result of post-European settlement land-uses including logging, burning, and grazing during the mid 1800s. This unique savanna structure was maintained throughout the early years of the Southern Illinois University but was radically altered when the University enacted a "protection" policy eliminating natural disturbances, particularly fire, which would have helped sustain the savanna habitat.

Thompson Woods provides an important lesson for land managers in making clear the distinction between preservation and conservation. Most natural area preserves are set aside to protect unique habitats such as glade barrens, hill prairies, or oak savannas. Often, these systems require periodic fire to maintain the unique habitat qualities and within the literal translation of "preservation," no active management is deemed the appropriate course and eventually, the set-aside preserve changes into a different system and the original unique habitat is no longer preserved, but lost. We hope the lessons of this University forest stand provide answers for those land managers responsible for maintaining such remnant forests on their properties.

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NETTIE HART MEMORIAL WOODLAND, ILLINOIS: PRESENT COMPOSITION AND CHANGES IN COMMUNITY STRUCTURE

Jeremy Shafer and John M. Edgington[1]

ABSTRACT.—An inventory of the woody vegetation of Nettie Hart Memorial Woodland in Champaign County, Illinois, has been conducted at approximately 10-year intervals from 1965 to 1995. Over this 30-year span, white oak, black oak, and northern red oak remained the most prominent components of the overstory community. At present, white oak and black oak are the major dominants. Northern red oak is decreasing in importance, while silver maple, American elm, slippery elm, and black cherry have all gained prominence. The hickory species have remained relatively stable since 1965.

Hart Memorial Woodland is a 13-ha woodland located along the Sangamon River near Mahomet in Champaign County, Illinois. The woodland was probably selectively harvested sometime during the period 1834 to 1854 (Johnson and Bell 1975). Since that time the woodland has re-mained relatively undisturbed. Hart Memorial Woodland is a xerophytic upland streamside forest within the Prairie Peninsula of east-central Illinois as described by Braun (1950). Stream systems were the major migratory pathways of woody species into Illinois prairies after glaciation (Gleason 1923).

There are two general types of upland forests in east-central Illinois: those occurring along major streams and rivers, and those extending into the prairie or isolated from other main bodies of timber. Upland sites bordering streams in east-central Illinois have been occupied by forest vegetation for a sufficient period of time to allow the development of alfisols distinct from the predominant mollisols (Boggess and Geis 1967). In other central Illinois forests, oak and hickory species are being replaced by the more shade tolerant mesic species such as sugar maple (*Acer saccharum* Marsh.) (Boggess and Geis 1967, Boggess and Bailey 1962, Edgington 1991).

Hart Memorial Woodland is unique regionally in that it is one of only a few that has been regularly inventoried since 1965 when acquired by the University of Illinois. This has provided a direct means of documenting vegetational changes through repeated sampling of permanent plots. The first inventory of Hart Memorial Woods was completed in 1965 (Root and others 1971). The second inventory was completed in 1977 (Johnson and others 1978). The third inventory was completed by Edgington in 1987 (unpublished). The most recent inventory of the woodland was completed in 1995 and will be used to describe the current species composition and structure and contrast these stand characteristics with those of the past.

STUDY AREA

Hart Memorial Woodland is located at 40º 14′ north latitude and 88º 21′ west longitude, Champaign County, IL. Elevation change is 9.4 m from the bottomland to the upland with a general slope of 30 degrees in the transition. The Sangamon River runs along the east border of the woodland and is subject to periodic flooding. Two small intermittent streams intersect the woodland.

The uplands of Hart Memorial Woodland consist mainly of Alfisols that developed in loess under the influence of forest vegetation. These soils are moderately well drained to well drained. The bottomland soils consist of Mollisols developed in stratified loamy alluvium on flood plains or low terraces of large streams under prairie

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vegetation. These soils are somewhat poorly drained to poorly drained (Mount 1982).

METHODS

Sampling and analysis procedures follow those of Root and others (1971). The forest is divided into fifty-two 50 m^2 (0.25 hectare) permanently marked blocks. Each of the 52 blocks is divided into quadrants by extending diagonal lines from the permanent corner markers resulting in a total of 208 plots. In each quadrant, the diameter of each stem greater than 6.4 cm diameter at breast height (dbh) was measured to the nearest 2.5 cm and recorded by species. The dbh of dead standing trees was also tallied and identified when possible.

Nested, square subplots of 5 m^2 and 2 m^2 were systematically located in quadrant 1 of each of the 52 blocks. Stems greater than 1.3 cm dbh (saplings) were measured in the 5 m^2 subplot and recorded by species and diameter classes of 2.5 cm and 5.0 cm. In the 2 m^2 subplot, all stems less than 1.3 cm dbh (seedlings) were measured and recorded by species. Height classes of less than 30.5 cm and greater than 30.5 cm, were used to differentiate the seedling sizes.

Frequency, density, and basal area were calculated for each species. Seedling and sapling frequencies were expressed as percentages of occurrence on 52 sample plots. Importance values for all tree species were developed from the sum of the relative density and relative basal area (McIntosh 1957). Nomenclature follows Mohlenbrock (1975).

RESULTS

A total of 31 woody species were tallied in 1995. These 31 species accounted for a total of 7,484 trees or 576 stems/hectare. The top 13 species are listed in table 1, along with their density and basal area, by diameter class and are ranked by their importance value.

The total basal area of the woodland was found to be 402.74 m^2 or 30.98 m^2/ha. White oak (*Quercus alba* L.), black oak (*Quercus velutina* L.), and northern red oak (*Quercus rubra* L.) accounted for 68 percent of the total basal area. White oak ranked first in importance value and accounted for 148.51 m^2 or 36 percent of the total basal area (table 1). The majority of this (65 percent) came from the 31 to 60 cm diameter class. Black oak ranked second in importance value followed by northern red oak. Silver maple (*Acer saccharinum* Marsh.) ranked fourth in importance value and in basal area with

Table 1.—*Subplot densities and percent frequencies of Hart Memorial Woodland, 1995*

Common name	Seedlings <1.27cm		Saplings >1.27cm	
	#/ha	Freq.	#/ha	Freq.
White oak	47	2	15	4
Slippery elm	2,740	38	109	19
American elm	1,586	8	146	21
Green ash	289	8	7	2
Black cherry	482	8	69	13
Sassafras	193	4	15	4
Hackberry	0	0	15	4
Bitternut hickory	961	25	7	2
Mockernut hickory	96	4	62	10
Shagbark hickory	47	2	30	6
Boxelder	47	2	0	0
Eastern redbud	0	0	7	2
Shingle oak	0	0	7	2
Dogwood spp.	193	6	7	2
Other	1,154	8	0	0
Total	7,836		497	

35.58 m^2. The species was well represented in all diameter classes.

Slippery elm (*Ulmus rubra* L.) and American elm (*Ulmus americana* L.) ranked fifth and sixth in importance value and accounted for 33.50 m^2 of basal area collectively (table 1). The majority of this was attributable to trees in the smaller diameter classes (7 to 15 cm and 16 to 30 cm). Green ash (*Fraxinus pennsylvanica* Marsh.) ranked seventh in importance value followed by black cherry (*Prunus serotina* Ehrh.). The majority of the green ash trees were in the 7 to 15 cm and 16 to 30 cm diameter classes with the majority of black cherry being in the 7 to 15 cm diameter size class.

The seedling density of all species combined was 7,836 seedlings/hectare and sapling density was 497 saplings/hectare (table 2). The elm species accounted for 55 percent of the seedling density and 51 percent of the sapling density. Bitternut hickory (*Carya cordiformis* Wang.) accounted for 12 percent of the seedlings. Black cherry accounted for 14 percent of the sapling density. Other seedlings and saplings were green ash, sassafras (*Sassafras albidum* (Nutt.) Nees.), dogwood (*Cornus* spp.), mockernut hickory (*Carya tomentosa* Poir.), and shagbark hickory (*Carya ovata* Mill.). Boxelder (*Acer negundo* L.) occurred only in the seedling size class and hackberry (*Celtis occidentalis* L.) was only in the sapling size class. Of the oak species, only white oak was in the seedling and sapling size class.

Table 2.—*Number of trees, basal area (m^2), and importance values by diameter class of Hart Memorial Woodland, 1987-1995*

Diameter class (cm)	7 to 15		16 to 30		31 to 60		61 to 90		>91		Total		IV
	No.	BA	No.	BA	No.	BA	No.	BA	No.	BA	No.	BA	
1995 SURVEY:													
White oak	37	0.52	328	16.31	595	97.08	89	33.84	1	0.77	1,050	148.51	37.0
Black oak	1	0.00	6	0.37	246	51.26	85	32.65	2	1.81	340	86.10	21.4
Northern red oak	8	0.07	5	0.27	56	11.44	63	25.94	3	2.20	135	39.92	9.9
Silver maple	290	2.71	164	6.54	68	9.86	28	11.91	5	4.56	555	35.58	8.9
Slippery elm	1,698	14.24	165	5.73	14	1.79	0	0.00	0	0.00	1,877	21.77	5.7
American elm	870	6.04	81	3.27	19	2.42	0	0.00	0	0.00	970	11.73	3.0
Green ash	59	0.55	46	2.03	27	4.49	6	2.35	0	0.00	138	9.42	2.4
Black cherry	664	5.26	72	2.34	14	1.76	0	0.00	0	0.00	750	9.37	2.4
Black walnut	47	0.51	37	1.76	31	4.94	2	0.74	0	0.00	117	7.95	2.0
Sassafras	713	5.12	23	0.72	0	0.00	0	0.00	0	0.00	736	5.84	1.5
Hackberry	95	0.84	37	1.47	14	2.36	3	1.12	0	0.00	149	5.78	1.5
Bitternut hickory	53	0.45	20	0.94	19	2.68	2	0.71	0	0.00	94	4.78	1.2
Bur oak	9	0.07	0	0.00	3	0.59	5	2.35	0	0.00	17	3.01	0.7
All other species[1]	446	3.50	73	2.95	33	5.03	4	1.51	0	0.00	556	12.99	3.3
1995 Totals	4,990	39.89	1,057	44.70	1,139	195.70	287	113.12	11	9.34	7,484	402.74	
1987 SURVEY:													
White oak	61	0.85	418	23.32	631	100.68	69	25.44	1	0.69	1,180	148.20	37.5
Black oak	2	0.02	9	0.56	316	63.28	70	26.88	0	0.00	397	90.74	22.9
Northern red oak	9	0.07	10	0.49	92	18.22	59	23.72	1	0.69	171	43.19	10.9
Silver maple	366	3.34	126	5.07	62	9.86	21	8.76	3	2.61	578	29.64	7.5
Slippery elm	1,622	12.38	95	3.22	9	0.91	2	0.77	0	0.00	1,728	17.28	4.6
Green ash	101	0.87	51	2.14	28	4.32	10	3.82	0	0.00	190	11.14	2.8
American elm	708	4.90	67	2.88	16	1.76	0	0.00	1	0.77	792	10.20	2.7
Black cherry	712	5.18	54	1.85	14	1.87	0	0.00	0	0.00	780	8.90	2.3
Black walnut	59	0.55	31	1.22	32	4.76	3	1.26	0	0.00	125	7.79	2.0
Sassafras	852	5.54	21	0.66	0	0.00	0	0.00	0	0.00	873	6.20	1.7
Hackberry	101	0.87	32	1.30	14	2.08	2	0.71	0	0.00	149	4.95	1.3
Bitternut hickory	33	0.27	19	0.84	12	1.86	1	0.37	0	0.00	65	3.34	0.9
Bur oak	1	0.01	5	0.22	3	0.45	5	2.36	0	0.00	14	3.04	0.8
All other species	349	2.84	69	2.76	37	5.52	3	1.15	0	0.00	458	12.26	3.1
1987 Totals	4,976	37.67	1,007	46.53	1,266	215.55	245	95.05	6	51.37	7,500	396.87	
1977 SURVEY:													
White oak	102	1.47	595	29.16	601	96.34	49	19.50	1	0.79	1,348	147.27	40.8
Black oak	4	0.03	7	0.42	266	50.90	53	19.43	0	0.00	330	70.78	19.5
Northern red oak	4	0.03	37	2.13	203	35.88	56	23.31	1	0.88	301	62.22	17.2
Silver maple	212	1.94	68	2.81	52	9.67	11	4.60	4	3.92	347	22.94	6.4
Slippery elm	864	5.96	59	2.23	9	1.33	1	0.48	0	0.00	933	10.01	2.9
Green ash	29	0.29	18	0.90	29	5.79	4	1.47	0	0.00	80	8.44	2.3
Black cherry	490	3.34	34	1.46	14	1.82	0	0.00	0	0.00	538	6.62	1.9
American elm	399	2.82	52	1.93	6	0.79	0	0.00	0	0.00	457	5.54	1.6
Mockernut hickory	36	0.33	24	1.08	20	2.91	1	0.34	0	0.00	81	4.65	1.3
Black walnut	25	0.22	9	0.49	25	3.37	1	0.34	0	0.00	60	4.42	1.2
Hackberry	89	0.80	26	1.11	13	1.69	1	0.36	0	0.00	129	3.98	1.1
Shagbark hickory	105	0.80	32	1.32	9	1.00	0	0.00	0	0.00	146	3.12	0.9
Sassafras	377	2.15	19	0.63	0	0.00	0	0.00	0	0.00	396	2.78	0.8
All other species	166	1.45	28	1.15	31	4.73	6	2.53	1	0.84	232	10.70	3.0
Total	2,902	21.63	1,008	46.81	1,278	216.22	183	72.37	7	6.43	5,378	363.47	

[2] See Appendix 1 for a list of "all other species" in Hart Memorial Woodland, 1995.

Shingle oak (*Quercus imbricaria* Michx.) was present in the sapling size class.

The 1995 inventory recorded 339 dead standing trees, or 26 stems/hectare. American elm and slippery elm accounted for 47 percent of the dead standing trees. The majority of these trees were in the 7 to 15 cm and 16 to 30 cm diameter classes.

DISCUSSION

Prior to 1995 three other inventories were conducted at Hart Memorial Woods. Raw data from the 1965 inventory were not available so comparisons were made with the inventory values derived by Root and others (1971). Table 1 lists importance values of each diameter class of the top 13 ranked species since the 1977 inventory. The species are ranked from highest to lowest by importance values from each inventory.

White oak, black oak, and northern red oak have had the highest importance values in all inventories (fig. 1). They have remained the dominant species with a combined basal area of 274.53 m^2 in 1995. This is similar to the 245.35 m^2 of basal area in 1965 measured by Root and others (1971) as well as the 280.27 m^2 measured in 1977 by Johnson and others (1978) and the 282.13 m^2 of basal area obtained by Edgington in 1987 (unpublished).

In the last three inventories, white oak accounted for over 50 percent of the total basal area. The majority of this basal area was in the 31 to 60 cm diameter class for all three inventories. Although white oak has decreased in number of stems in the 7 to 15 cm and 16 to 30 cm diameter classes, it remains the only oak species well represented in these diameter classes. White oak has increased in number of stems and basal area in the 61 to 90 cm diameter class due to recruitment from the 31 to 60 cm diameter class.

Black oak accounted for 86.1 m^2 of basal area in 1995; the majority of this basal area is in the 31 to 60 cm diameter class. However, compared to 1977, black oak has decreased in number of stems in the 31 to 60 cm diameter class, but has increased in the 61 to 90 cm and 91+ cm diameter classes. Since 1977, black oak has increased in total basal area with a net gain of 15.32 m^2.

In 1995, northern red oak accounted for 39.92 m^2 of basal area. The majority of this basal area was in the 61 to 90 cm diameter class. Northern red oak has shown a steady decrease

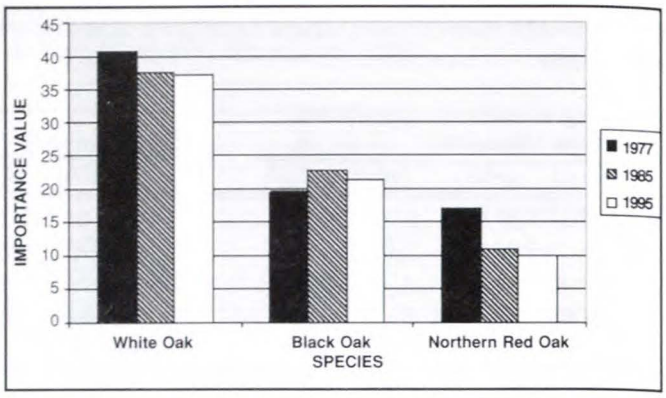

Figure 1.—*Three tree species with the highest importance values in Hart Memorial Woodland.*

in number of stems and basal area in all diameter classes except for the 61 to 90 cm and 91+ cm diameter classes. Since 1977, northern red oak has decreased in total number by more than 55 percent. White oak may begin to increase in importance due to its high number of total stems as the large northern red oaks and black oaks die.

Silver maple has increased in number of stems since 1977 with the greatest increase occurring in the 16 to 30 cm diameter class. The importance value of silver maple has remained fourth in rank, but it has increased by an average of 1.25 per inventory since 1977. Since total numbers have increased and 1.94 ha of Hart Woods is a bottomland woodland, silver maple is likely to continue its slow increase.

Slippery elm and American elm have maintained stable importance values in the smaller diameter classes although neither species has attained diameters greater than 61 cm due to high mortality of smaller individuals. It does not appear that either species will form an important component of the future overstory, presumably due to the prevalence of Dutch elm disease and phloem necrosis in the region and the consequent mortality of young trees. With the elm species comprising more than 50 percent of the seedling and sapling recruitment, it seems likely that they will have a negative effect on the regeneration of shade intolerant species such as the oaks and hickories.

The hickory species have maintained their importance, collectively, but have remained a small component of the overstory. They remain an important component in the smaller diameter classes, but only bitternut hickory has increased in importance in diameter classes above the 7 to 15 cm class. There are high numbers of

seedlings and saplings of hickory spp. so their importance may begin to increase in the smaller diameter classes. Unless more hickories grow to reach the middle diameter classes, any canopy gaps created will likely be filled by other species. The hickories will probably remain a minor component of the overstory for some time since there are only two hickories with a dbh greater than 61 centimeters.

Sassafras has maintained importance in the sapling stage and has increased since 1977 in the 7 to 15 cm diameter class, as well as the 16 to 30 cm diameter class. Sassafras is classified as intolerant of shade at all ages (USDA Forest Service 1965). The presence of sassafras suggests that the stand was more open at one time. Those sassafras that have established a position in the canopy are expected to maintain their position in the canopy. Due to its intolerance, sassafras recruitment is expected to be suppressed without the creation of canopy gaps.

Hart Memorial Woodland will likely continue to be dominated by white oak and black oak into the near future. Oak regeneration may continue to be suppressed lacking future disturbance. Northern red oak is declining in importance and may continue to do so due to poor seedling and sapling recruitment. Whether white oak will gain importance by filling canopy gaps created by mortality of large northern red oak and black oak can be predicted due to its greater longevity. Silver maple may increase in importance in the flood plain portion of the site as it continues to have high recruitment into larger diameter classes.

The current inventory indicates a continuing trend towards white oak and black oak dominance, which is in contrast to other nearby forests that show a shift towards more mesic species, especially sugar maple. The total absence of sugar maple from the forest is in marked contrast to the situation in other forests of east central Illinois as described by Boggess and Geis 1967, Boggess and Bailey 1962, and Edgington 1991. The reason for the absence of sugar maple in Hart Memorial Woodland is not known. Continued inventories will be necessary in order to further document species composition and changes in community structure of Hart Memorial Woodland.

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APPENDIX 1

Reference to "all other species" measured in Hart Memorial Woodland, 1995.

Common name	Scientific name
Mockernut hickory	*Carya tomentosa* (Poir.) Nutt.
Honeylocust	*Gleditsia triacanthos* L.
Shagbark hickory	*Carya ovata* (Mill.) K. Koch.
American basswood	*Tilia americana* L.
American sycamore	*Platanus occidentalis* L.
Boxelder	*Acer negundo* L.
Hawthorn spp.	*Crataegus* spp. L.
Eastern redbud	*Cercis canadensis* L.
Shingle oak	*Quercus imbricaria* Michx.
Ironwood	*Ostrya virginiana* (Mill.) K. Koch.
White ash	*Fraxinus americana* L.
Pawpaw	*Asimina triloba* L.
Mulberry spp.	*Morus* spp. L.
Kentucky coffeetree	*Gymnocladus dioicus* (L.) K. Koch.
Eastern cottonwood	*Populus deltoides* Marsh.
Osage-orange	*Maclura pomifera* L.
Ohio buckeye	*Aesculus glabra* Willd.
Dogwood spp.	*Cornus* spp. L.

USING ROC CURVES TO COMPARE NEURAL NETWORKS AND LOGISTIC REGRESSION FOR MODELING INDIVIDUAL NONCATASTROPHIC TREE MORTALITY

Susan L. King[1]

ABSTRACT.—The performance of two classifiers, logistic regression and neural networks, are compared for modeling noncatastrophic individual tree mortality for 21 species of trees in West Virginia. The output of the classifier is usually a continuous number between 0 and 1. A threshold is selected between 0 and 1 and all of the trees below the threshold are classified as mortality trees and all of the trees above the threshold are classified as survival trees. Selecting the threshold that has both a high sensitivity and specificity is a major decision. A receiver operating characteristic (ROC) curve graphically describes the performance of the classifier without the requirement of a threshold. Its accuracy is measured by the area under the curve (AUC). A neural network is the superior classifier because it has a higher AUC statistic.

The output from a statistical procedure, such as logistic regression or neural networks (Bishop 1995), is traditionally classified as to group membership, i.e., mortality or survival, based on whether the output is above or below a threshold or cutoff. The threshold is selected based on historical information, such as the survival or mortality rate of the population, or by chance (cutoff = 0.5). The comparison statistic is usually the correct number of matches, or overall accuracy, between the predicted and observed outcome.

The comparison statistic suffers from several disadvantages. First, different procedures perform better at different decision thresholds. For example, logistic regression may be superior over neural networks at low decision thresholds, whereas neural networks may be superior at higher thresholds. Second, in some applications there is a higher cost associated with different types of misclassifications. Third, this statistic is extremely sensitive to the prevalence of the classes in the data, being biased toward the majority class.

In any noncatastrophic tree-mortality model, the overwhelming population is survival trees. A classification technique could have a high overall accuracy and completely misclassify mortality trees because of their low prevalence. Another classification statistic, such as the kappa statistic or receiver operating characteristics (ROC), is required. This paper applies ROC analysis to determine the classification superiority of either neural networks or logistic regression for modeling individual tree mortality. The most common type of neural networks, a feed-forward backward propagation neural network, was selected for this study. A neural network is not restricted to a categorical dependent variable.

ROC CURVES

ROC curve analysis developed in electrical engineering in the 1950s to detect electromagnetic signals from noise. In the past 30 years, ROC curve analysis has been applied to problems in medical diagnosis and psychology. The diagnostic decision-making process is essentially the same across disciplines (Swets and others 2000).

Environmental applications of ROC curves are scarce and a goal of this paper is to introduce this topic to the forest resource community. One application is a land-cover change model (Pontius and Schneider 2001) and another environmental paper evaluated tree mortality following fire damage (Saveland and Neuenschwander 1990). Several medical imagining applications using neural networks are described in Salchenberger and others (1997)

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and Wu and others (1993). Other useful references and applications are found in Swets and Pickett (1982), Hanley and McNeil (1982), Gohagan and others (1984), and Swets (1996).

In any application, a threshold distinguishes between noise and a signal, survival and mortality, cancerous and non-cancerous cells. Anything below the threshold is classified as "noise" and anything above the threshold is classified as a "signal." If the threshold is too low, not only is it more likely that a "signal" will be detected, but also that "noise" will be mistaken for a "signal." Similarly, if the threshold is too high, it is very likely that a signal will be misclassified as noise.

At any threshold, a confusion matrix is generated by tabulating the true positives, the true negatives, and the two misclassifications (table 1). In this application, true positives are the number of trees that were predicted to die that actually died and true negatives are the trees that were predicted to survive that actually survived. False positives occur when the trees are predicted to die but actually survive. The opposite is a false negative. The overall accuracy is the sum of the true positives (M_{11}) and true negatives (M_{22}) divided by the total number of observations in the matrix (N).

In medical diagnostics, the rate of true positives (percent of positive events correctly classified) is called sensitivity, whereas the rate of true negatives (percent of nonevents correctly classified) is called specificity. Sensitivity is a measure of accuracy for predicting events, whereas specificity is a measure of accuracy of predicting nonevents. An ROC curve is a plot of the sensitivity versus one minus the specificity (1–specificity), false positives, for a locus of decision thresholds. Both the ordinate and abscissa range from 0 to 1.

The ROC curve for a perfect test would be an inverted "L", going up the ordinate from 0 to 1 and then traveling in a horizontal line from (0,1) to (1,1). A test that makes decisions by chance would have the 45-degree line as the ROC curve. The area under the curve (AUC) is calculated using the trapezoidal rule from calculus or another numerical integration technique and is a measure of the performance of the classifier. Each decision threshold is a point on the curve. A perfect test would have an AUC of 1, whereas a test by chance would have an AUC of 0.5. The AUC provides a measure of performance that is less sensitive to the prevalence of a class in the data and measures the performance of the classifier across the entire range of thresholds.

For logistic regression, the OUTROC option in PROC Logistic in SAS (SAS Institute, Cary, NC) provided both the AUC statistics and the locus of points for the ROC curve. For neural networks, a user written program in SAS provided the AUC curve statistics and the locus of points for the ROC curve.

INDIVIDUAL TREE MORTALITY

An important part of forest management is predicting the growth and yield of individual trees. Growth models have four components: survivor growth, ingrowth, cutting, and mortality.

Table 1.—*Classification results for a threshold*

		Observed: Mortality	Observed: Survival	Totals
Predicted	Mortality	True Positives (M11)	False Positives (M12)	M11 + M12 = N1
	Survival	False Negatives (M21)	True Negatives (M22)	M21 + M22 = N2
	Totals	M11 + M21	M12 + M22	N = M11 + M12 + M21 + M22

Predicting noncatastrophic individual tree mortality is the most difficult of the four components to model because an overwhelming number of trees survive from the first measurement to the second measurement. King and others (2000) evaluates neural networks, logistic regression, and two support vector methods to predict noncatastrophic individual tree mortality. They determined that neural networks were the superior technique for evaluating individual tree mortality. ROC curve analysis was one of their tools for reaching this conclusion.

DATA AND VARIABLES

The data for this project came from the 1964, 1975, and 1988 forest inventories of West Virginia collected by the USDA's Northeastern Forest Inventory and Analysis (NEFIA) unit. This analysis uses trees that were alive at the second measurement (1975) and were larger than 5 inches in diameter at breast height (d.b.h.). Some of the trees that were measured in the 1975 and 1988 inventories were not measured in the 1964 inventory. Their diameters were estimated using a linear regression procedure for estimating previous diameters developed by King and Arner (1999).

Table 2 shows the distribution of the species into 21 groups and the number of survival and mortality trees in each species group. The species groups were classified according to growth and competitive characteristics. The same set of independent variables was included in each species group. In West Virginia, there were no trees in species group 13, sweetbay.

For modeling, each species group was broken into a model and a validation data set. The modeling procedures require each pattern, or combination of dependent and independent variables, to be represented in the model data set. This is an impossible objective given several continuous independent variables. The solution is to obtain a proportional representation of the mortality by diameter class and hopefully a proportional representation of each pattern, the data were divided into 2-inch diameter classes. All trees 23 inches d.b.h. and larger were grouped into one class. Using a random number generator, approximately 75 percent of the trees in each diameter and mortality class were placed in the model data set and the remaining trees were placed in the validation data set. Mortality is highest in the lower and upper

Table 2.—*Species group assignments*

Species groups	Common names	Number of survival trees	Number of mortality trees
1	Red spruce	107	31
2	Eastern redcedar, shortleaf pine, table mountain pine, pitch pine, Eastern white pine	283	113
3	Virginia pine	495	203
4	Eastern hemlock	304	15
5	Boxelder, sugar maple	908	94
6	Red maple	1298	124
7	Yellow birch, sweet birch	70	193
8	Hickory	124	219
9	American beech	1031	167
10	White ash, green ash	31	62
11	Butternut, black walnut	145	48
12	Yellow-popla	1344	140
13	Sweetbay	0	0
14	Black cherry	44	62
15	White oak, chestnut oak, post oak Scarlet oak, Southern red oak, overcup oak, chinkapin oak, pin oak,	3635	423
16	Willow oak Willow oak	541	129
17	Northern red oak, black oak	2076	320
18	Black locust	335	226
19	American basswood	355	43
20	All other commercial species	761	203
21	All other noncommercial species	432	306

diameter classes. Smaller trees are more likely to die from competition, whereas trees that are larger, and therefore older, suffer from cumulative life stresses.

The models require three time periods of data. The independent variables are derived from the variables collected in the first two measurements. The dependent variable, mortality, is from the third measurement. Many possible independent variables were investigated. Variables such as crown class and crown ratio were not available for all of the trees in the second measurement. Modeling of the missing crown classes and crown ratios was not successful.

In many other individual tree mortality models, one of the independent variables is site quality. Site index, the height of a free-growing dominant or codominant tree at an index age, is used to measure site quality. Most of the site-index equations are based on the assumption that the trees on the plot or in a stand are of the same species and age. These assumptions are violated in most of the Northeastern states. From NEFIA's experience, site is not a useful predictor of growth and slow growth is an indicator of susceptibility to mortality.

Six independent variables were included in the final model: basal area at the second inventory (BA2); basal area increment (BAI); relative basal area increment (RBAI); basal area larger than the subject tree (BAL); the interaction of basal area and basal area increment (BABAI); and soundness. Guan and Gertner (1991) used d.b.h., d.b.h. increment, the interaction between d.b.h. and d.b.h. increment, and soundness as independent variables in their comparison of logistic regression and neural networks for modeling individual tree mortality. In addition to an expanded variable list, three time periods of data were used instead of two.

BAI is the change in the basal area of an individual tree between the first and second inventories divided by the number of years between NEFIA plot measurements. It is a measure of the vigor of the tree and is indicative of its survival. A tree with low vigor is more likely to die.

RBAI is the ratio of each individual tree's BAI with the total BAI of all of the trees on the plot. RBAI is a measure of an individual tree's vigor as compared to the vigor of the other trees on the NEFIA plot.

BAL measures the competitive position of an individual tree within a plot as expressed by the sum of the basal area of all trees larger than the subject tree. The largest tree on the plot has a BAL of zero and the smallest tree has a BAL equal to the sum of the basal areas of the larger trees. Teck and Hilt (1990) report that BAL had a higher correlation with survival than the ratio of d.b.h. to quadratic mean stand diameter and the ratio of tree basal area to plot basal area. Also King and Arner (1999) found that BAL was a better explanatory variable for competition than either relative diameter or relative quadratic diameter.

BABAI captures the relationship between tree size and vigor. Buchmann and others (1983) postulate that the survival rate for low vigor trees depends on tree size and that the survival peaks between the juvenile stage and senescence. The relationship between species and size is species dependent. Long-lived trees with low vigor are more likely to survive if they are past the sapling stage. Short-lived trees with low vigor are more likely to survive when they are small.

Soundness is an estimated variable. It is a measure of rot and decay in the tree. Not all of the information was available for every tree in the second measurement. Merchantability (MRCH2) is a class variable that captures cull and form of the tree bole. Trees of poor form and high cull die more frequently. The mean of the soundness calculations for the third measurement was assigned to a tree based on its MRCH2 classification. The following values were assigned to soundness.

MRCH2	Soundness
1	99
2	98
3	81
5	90
6	45

Growing stock trees have a MRCH2 of 1, 2, or 3 and cull trees have a MRCH2 of 5 or 6. Trees with a MRCH2 of 1 have no cull, whereas trees with a MRCH2 of 2 or 5 have only sound cull and trees with a MRCH2 of 3 or 6 have rotten cull. A frequency analysis between mortality and cull shows that mortality occurs more frequently if the cull is rotten.

RESULTS AND CONCLUSIONS

The ROC curves for the model and validation data sets were created and analyzed for each species group. Each curve has 100 points, each corresponding to a threshold. The population mortality ranged from 4.5 percent for eastern hemlock to 42.2 percent for all other noncommercial species. The model data set is used to

establish the cutoff. However, the superiority of a technique is determined by the model's ability to classify new data. This is evaluated by the validation data set. The results show a variety of ROC curves. By definition all of the curves start at the origin and end at (1,1). At the origin, all of the survival trees are perfectly classified and all of the mortality trees are misclassified, whereas at (1,1) all of the mortality trees are perfectly classified.

The lower cutoffs are associated with a low sensitivity, a low 1-specificity, and are biased toward classifying survival trees correctly. The higher cutoffs are associated with higher sensitivity values and are biased toward correctly predicting mortality at the expense of survival trees. The overall accuracy is high at low cutoffs and is low at the higher cutoffs due to the large number of survival trees. Landis and Koch (1977) classified different ranges of kappa. These classifications are poor ($\kappa \leq 0.4$), good ($0.4 < \kappa \leq 0.75$), and excellent ($\kappa > 0.75$).

Since the purpose of this paper is to illustrate the technique, only the results for three species are presented: black cherry, Virginia pine, and yellow-poplar. For species group 14, black cherry, figure 1a shows neural networks trading positions of superiority with logistic regression over the decision space. Overall, neural networks are slightly superior with an AUC of 0.852 to an AUC of 0.809 for logistic regression. Since it is difficult to discern the changes in the graph, Appendix table 1 presents selected thresholds used to create the graphs. From Appendix table 1 and from figure 1a, the two procedures are equivalent until a threshold of 0.06. The overall accuracy is high, over 88 percent, yet the kappa statistic is low, less than 0.01, indicating a lack of agreement between the two classifiers. From a threshold of 0.06 to 0.18, logistic regression is superior as indicated by the larger sensitivity. Both procedures have the same 1–specificity value.

It is difficult to discern the difference graphically, but the logistic regression curve is slightly to the left of the neural network curve in figure 1a. Both procedures are classifying survival the same, but logistic regression correctly classifies more trees that actually died. From a threshold of 0.18 to a threshold of 0.90, neural networks are superior to logistic regression. The latter threshold can be found from the graph and Appendix table 1. The two curves cross at approximately a sensitivity of 0.84 and a 1-specificity of 0.38. This corresponds to a cutoff between 0.90 and 0.92 in Appendix table 1. At a threshold of 0.92, the overall accuracy is 58.5 percent for neural networks and 64.8 percent for logistic regression. The procedures swap positions of superiority twice again before the final cutoff of 1.0. The highest overall number of correct matches for neural networks and logistic regression occur at thresholds 0.64 and 0.44 to 0.46, respectively. The highest kappa for neural networks is 0.523, which occurs at a threshold of 0.68, and the maximum kappa for logistic regression is 0.472, which occurs at a threshold of 0.72.

Whether the optimal threshold is from the maximum value of the kappa statistic, maximum overall accuracy, by chance, or another criterion, this threshold is found from the model data set and applied to the validation data set. The validation data set has an AUC of 0.745 for

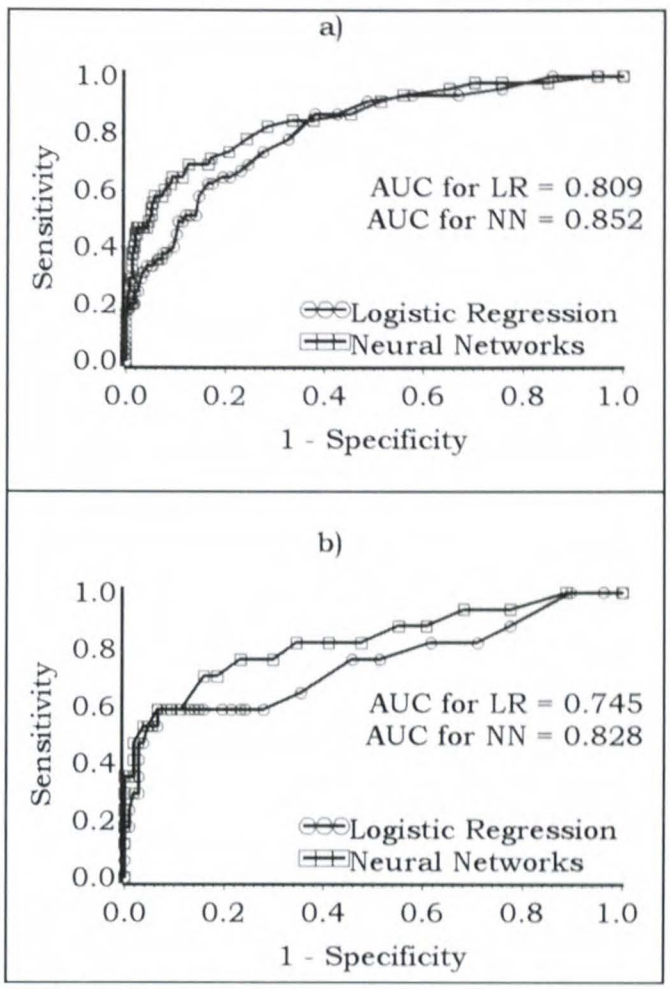

Figure 1.—*Black cherry ROC curves for: a) the model data; and b) the validation data set. Of the 378 trees in the model data set, 45 were mortality trees and of the 124 trees in the validation data set, 17 were mortality trees. Neural networks are the superior classifier due to the higher AUC statistics. Each point on the curve represents a threshold.*

logistic regression and 0.828 for neural networks. Neural networks and logistic regression are equivalent until a sensitivity of 0.12, which occurs at a threshold of 0.06. Logistic regression is superior until a threshold between 0.18 and 0.20, which corresponds to a sensitivity of around 0.2. The two procedures are close with neural networks having the edge until a sensitivity of 0.6, which corresponds to a threshold of around 0.86. Neural networks are superior to logistic regression over the remaining thresholds. For any of the optimal cutoffs from the model data set, neural networks would be the superior procedure at that threshold.

Virginia pine is an example of an almost perfect classifier (fig. 2). The AUC for neural network and logistic regression model data sets is 0.941 and 0.937, respectively. This indicates that neural networks are a slightly better classifier and that both procedures are almost perfect classifiers. The AUC for the validation data sets is 0.917 for neural networks and 0.911 for logistic regression. The curves have an inverted "L" shape, with high sensitivity and low 1–specificity values.

Conversely, yellow-poplar is an example of a poor classifier (fig. 3). For both the model and validation data sets, the AUC statistics are between 0.7 and 0.8. Although, these statistics are above 0.5, they are still quite low. The graphs in figures 3a and 3b rise slowly after a sensitivity of 0.3.

FURTHER APPLICATIONS

In many applications, especially in medical diagnostics, there is a greater cost associated

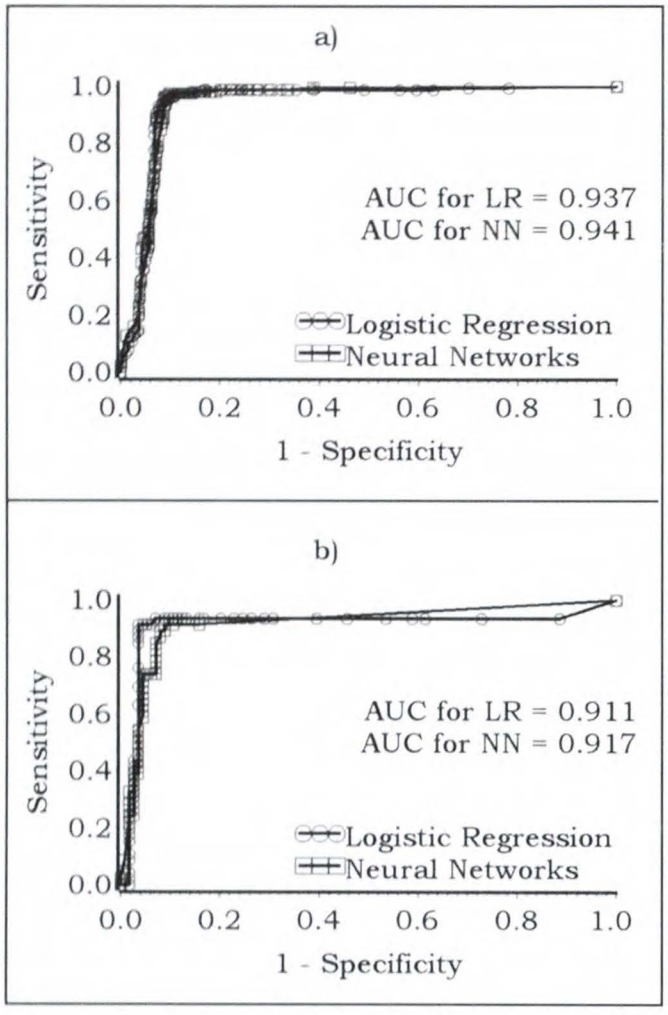

Figure 2.—*Virginia pine ROC curves for: a) the model data set; and b) the validation data set. Of the 538 trees in the model data set, 157 were mortality trees and of the 160 trees in the validation data set 46 were mortality trees. Both classifiers are equivalent and their inverted "L" shape curve indicates that they are close to perfect.*

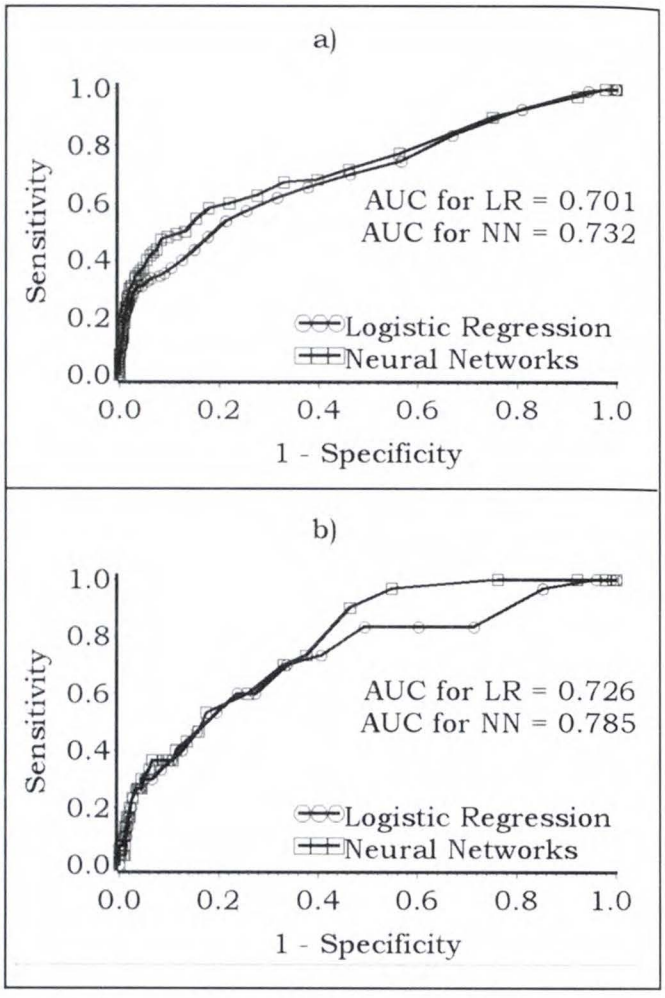

Figure 3.—*Yellow-poplar ROC curves for: a) the model data set; and b) the validation data set. Of the 1,122 trees in the model data set, 110 were mortality trees and of the 362 trees in the validation data set, 30 were mortality trees. Although neural networks are superior, both procedures are poor classifiers as indicated by their closeness to a 45-degree line.*

with false negatives than false positives. A cost or a loss matrix for each of the classes is defined in a confusion matrix. A risk function at each cutoff is defined similarly to that proposed by Bishop (1995) by combining the confusion matrix, cost or loss matrix, and the prior classification probabilities. This measurement is still sensitive to the prior probabilities of class occurrence and the decision threshold. A series of thresholds between 0 and 1 is selected and the risk is calculated at each threshold. The optimal cutoff corresponds to the minimum risk. Mathematically risk is defined as:

$$R=\sum_{j=1}^{c} R_j P(C_j) \qquad (1)$$

where

$$R_j=\frac{1}{N_j}\sum_{k=1}^{c} M_{jk} L_{jk}$$

M_{jk} = number of trees in the confusion matrix at row j and column k

L_{jk} = loss associated with misclassifying a tree

$P(C_j)$ = prior probability that a tree comes from class j

N_j = is the number of trees in class j

c = the number of classes which is equal to j or k

Unlike medical diagnostics, it is difficult in forestry to assign an economic value for loss or even determine which is a more severe misclassification, a false positive or a false negative. In medical diagnostics, it is obviously more costly to make a false negative misclassification error than a false positive misclassification error. With the latter, further testing should reveal the error, whereas no further testing is usually performed in the former case.

In forestry, if the purpose of classification were to estimate volume for an entire state, then the number and size of the misclassified trees would either inflate or deflate the estimates. It is not clear which type of error would be worse. A timber harvester would incur cost of locating the tree on the ground and lost revenue for trees that were predicted to survive but died, a false negative. For a false positive, there would be a missed opportunity cost of harvesting the tree. There would be no loss incurred for correctly classifying the tree.

For demonstration purposes, assume that the loss matrix is given in table 3. There is no loss in correctly classifying mortality or survival. A false negative has a slightly higher loss than a false positive. Using equation 1, risk is calculated for every threshold value for both logistic regression

Table 3.—*Loss matrix for risk evaluation*

		Observed	
		Mortality	Survival
Predicted	Mortality	$0.00	$600.00
	Survival	$700.00	$0.00

and neural networks. The prior probability for black cherry mortality, 12.35 percent, is taken to be the total number of mortality trees in the sample population, the combined model and validation data sets. Similarly, the prior probability for survival was found and was 87.65 percent. The minimum risk for neural networks was $57.50 and occurred at a threshold of 0.64. The minimum risk for logistic regression was found to be $71.09, occurring at a cutoff of 0.44. These cutoffs would be applied to the validation data set.

CONCLUSIONS

ROC curve analysis allows the user to compare two or more classifiers over the range of thresholds and to visually compare the performance of the classifiers. In most cases, neural networks were superior to logistic regression. The two procedures were generally equivalent at the lower threshold values and neural networks gained its superiority at the higher threshold values. Risk analysis is a useful tool for finding the optimal threshold. If there were a sufficient number of trees, breaking the models into three categories of young, middle-, and older-age trees might improve the results.

ROC curve analysis has many environmental applications and has been underutilized. It is a useful tool for comparing maps produced by different classifiers. The ROC methodology presented above is limited to "yes" or "no" class membership. Extension of the dichotomous ROC curve analysis to multiple cases is a current area of research (Dreiseitl and others 2000, Pontius and Schneider 2001).

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Appendix Table 1.—*Selected thresholds for black cherry for logistic regression (LR) and neural networks (NN)*

	Model				Validation			
Thres-hold	LR	% correct kappa sensitivity 1-specific.	NN	% correct kappa sensitivity 1-specific.	LR	% correct kappa sensitivity 1-specific.	NN	% correct kappa sensitivity 1-specific.
0.00	0 45 0 333	88.1 0.00 0.00 0.00	0 45 0 333	88.1 0.00 0.00 0.00	0 17 0 107	86.3 0.00 0.00 0.00	0 17 0 107	86.3 0.00 0.00 0.00
0.02	1 44 0 333	88.4 0.039 0.02 0.00	1 44 0 333	88.4 0.039 0.02 0.00	0 17 0 107	86.3 0.00 0.00 0.00	0 17 0 107	86.3 0.00 0.00 0.00
0.04	1 44 0 333	88.4 0.039 0.02 0.00	1 44 0 333	88.4 0.039 0.02 0.00	0 17 0 107	86.3 0.00 0.00 0.00	0 17 0 107	86.3 0.00 0.00 0.00
0.06	2 43 0 333	88.6 0.076 0.04 0.00	1 44 0 333	88.4 0.039 0.02 0.00	2 15 0 107	87.9 0.187 0.12 0.00	0 17 0 107	86.3 0.00 0.00 0.00
.	.	.	.	.	.	.	.	.
.	.	.	.	.	.	.	.	.
.	.	.	.	.	.	.	.	.
0.18	6 39 0 333	89.7 0.213 0.13 0.00	10 35 1 332	90.5 0.326 0.22 0.00	3 14 0 107	88.7 0.27 0.18 0.00	3 14 0 107	88.7 0.27 0.18 0.00
.	.	.	.	.	.	.	.	.
.	.	.	.	.	.	.	.	.
.	.	.	.	.	.	.	.	.
0.50	8 37 1 332	89.9 0.267 0.18 0.00	17 28 5 328	91.3 0.466 0.38 0.02	3 14 1 106	87.9 0.25 0.18 0.01	6 11 0 107	91.1 0.485 0.35 0.00
.	.	.	.	.	.	.	.	.
.	.	.	.	.	.	.	.	.
.	.	.	.	.	.	.	.	.
0.68	15 30 17 316	87.6 0.323 0.33 0.05	21 24 8 325	91.5 0.523 0.47 0.02	5 12 3 104	87.9 0.342 0.29 0.03	6 11 1 106	90.3 0.457 0.35 0.01
.	.	.	.	.	.	.	.	.

Threshold	Model LR	Model % correct / kappa / sensitivity / 1-specific.	Model NN	Model % correct / kappa / sensitivity / 1-specific.	Validation LR	Validation % correct / kappa / sensitivity / 1-specific.	Validation NN	Validation % correct / kappa / sensitivity / 1-specific.
0.72	16 29 24 309	86.0 0.472 0.36 0.07	21 24 12 321	90.5 0.487 0.47 0.04	8 9 3 104	90.3 0.52 0.47 0.03	7 10 2 105	90.3 0.490 0.41 0.02
0.88	30 15 77 256	75.7 0.273 0.67 0.23	35 10 81 252	75.9 0.318 0.78 0.24	10 7 23 84	75.8 0.267 0.59 0.21	12 5 17 90	82.3 0.422 0.71 0.16
0.90	33 12 92 241	72.5 0.258 0.73 0.28	38 7 112 221	68.5 0.253 0.84 0.34	10 7 26 81	73.4 0.235 0.59 0.24	13 4 25 82	76.6 0.349 0.76 0.23
0.92	39 6 127 206	64.8 0.224 0.87 0.38	39 6 151 182	58.5 0.173 0.87 0.45	11 6 38 69	64.5 0.163 0.65 0.36	14 3 37 70	67.7 0.259 0.82 0.35
0.94	41 4 162 171	56.1 0.169 0.91 0.49	42 3 186 147	50.00 0.136 0.93 0.56	13 4 55 52	52.4 0.111 0.76 0.51	14 3 51 56	56.5 0.159 0.82 0.48
0.96	42 3 223 110	40.2 0.085 0.93 0.67	44 1 234 99	37.8 0.085 0.98 0.70	14 3 76 31	36.3 0.04 0.82 0.71	15 2 65 42	46.0 0.107 0.88 0.61
0.98	45 0 286 47	24.34 0.038 1.00 0.86	44 1 283 50	24.9 0.034 0.98 0.85	17 0 96 11	22.6 0.03 1.00 0.90	16 1 83 24	32.3 0.055 0.94 0.78
1.00	45 0 333 0	11.9 0.00 1.00 1.00	45 0 333 0	11.9 0.00 1.00 1.00	17 0 107 0	13.7 0.00 1.00 1.00	17 0 107 0	13.7 0.00 1.00 1.00

MODELING HARDWOOD CROWN RADII USING CIRCULAR DATA ANALYSIS

Paul F. Doruska, Hal O. Liechty, and Douglas J. Marshall[1]

Cylindrical data are bivariate data composed of a linear and an angular component. One can use uniform, first-order (one maximum and one minimum) or second-order (two maxima and two minima) models to relate the linear component to the angular component. Crown radii can be treated as cylindrical data when the azimuths at which the radii are measured are also recorded. The radius is the linear component, while the azimuth is the angular component. Crown radii were measured as part of a hardwood forest research project at Arkansas Post National Memorial near Gillett, AR. The forest is composed of various stands of upland and bottomland hardwoods. Basic descriptive statistics, and uniform, first-order and second-order models were used to examine the circular patterns of crown radii by species.

A total of 167 trees of various sizes were used in this analysis: 48 sweetgum (*Liquidambar styraciflua* L.), 42 southern red oak (*Quercus falcata* Michx.), 41 American elm (*Ulmus americana* L.), 26 willow oak (*Quercus phellos* L.), and 10 water oak (*Quercus nigra* L.). The crown radii (meters) were measured in the north, northeast, east, southeast, south, southwest, west, and northwest directions and were made relative by dividing each by the respective total tree height in meters prior to use in the analysis. American elm, on average, possessed the largest relative crown radii (23 percent of tree height) whereas sweetgum possessed the smallest (10 percent of tree height). Tree to tree variation in relative crown radii was largest, on average, in willow oak and lowest in sweetgum as determined by standard deviations (13 percent of tree height and 5 percent of tree height, respectively). Within tree variation in relative crown radii was largest in willow oak, on average, and smallest in American elm as determined by the within tree coefficient of variation (64 percent and 41 percent, on average, respectively).

The first-order cylindrical model was significant (at $\alpha = 0.10$) for 30 percent of the willow oak; about 20 percent of the southern red oak, water oak, and American elm; and 10 percent of the sweetgum trees measured. A significant first-order model implies that the crown radii were largest in one particular direction (azimuth) and then decreased to a minimum at another direction (azimuth). The second-order cylindrical model was significant (at $\alpha = 0.10$) for three willow oak (about 12 percent of those measured), two sweetgum, and one southern red oak. A significant second-order model implies that the crown radii were largest in two particular directions (azimuths) and then decreased to minimums at two other directions (azimuths). A uniform model (approximately the same crown radii in all directions) was assumed appropriate for the remaining trees. Of the species examined, willow oak appears to be quite opportunistic with respect to radial crown expansion whereas sweetgum appears to be the least opportunistic.

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SILVAH 5.1: DEVELOPING INTERIM GUIDELINES FOR MANAGING OAK IN PENNSYLVANIA THROUGH MULTI-AGENCY COOPERATION

Patrick Brose, Susan Stout, Gary Miller, and Kurt Gottschalk[1]

The SILVAH decision-support system developed by numerous researchers at the Forestry Sciences Lab in Irvine, PA, has a proven track record for prescribing appropriate silvicultural treatments for the cherry-maple and northern hardwood forests of the Allegheny Plateau region. However, it has had only limited usage in mixed-oak forests of the same region and virtually no usage in the mixed-oak forests of neighboring regions (Ridge and Valley, Blue Ridge Mountains, etc.) of Pennsylvania because it is not perceived to be applicable to these regions. In late 1999, the Northeastern Research Station, Pennsylvania Bureau of Forestry, Pennsylvania State University, Allegheny National Forest, and the private sector began a cooperative effort to make SILVAH more applicable and widely used.

Experienced oak foresters and researchers from these agencies initially met to review existing oak knowledge and place it within the SILVAH prescription framework. Four major hurdles were encountered:

1) how to evaluate the competitive ability of existing oak regeneration,
2) what constituted a plot stocked with adequate oak regeneration,
3) how to sequence appropriate silvicultural treatments to address the most likely management scenarios, and
4) how to account for wide differences in site conditions within Pennsylvania.

Oak regeneration guidelines developed in the Midwest and the Appalachians and the participant's field experience were combined to overcome each of the hurdles and to create interim guidelines.

At the beginning of the 2000 field season, Northeastern Research Station scientists conducted a series of 2-day training sessions for Bureau of Forestry personnel to explain the changes in the inventory and computer-entry procedures. These field foresters then tested the draft interim guidelines during the remainder of the year. Several minor problems were discovered, but overall the new SILVAH procedures worked well. A follow-up meeting of the original partners and the field foresters revised the draft interim guidelines by incorporating their findings and concerns into final interim oak management guidelines for Pennsylvania. These guidelines are now being programmed into SILVAH 5.1 and will be published as a Forest Service General Technical Report.

This cooperative multi-agency approach of combining existing guidelines with extensive field experience led to creation of usable guidelines in a short period of time relative to use of a research-only approach. It also identified critical knowledge gaps that have been prioritized and are being actively studied so future versions of SILVAH for oak will be based as solidly in research as the current version of SILVAH for cherry-maple and northern hardwood forests.

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DIFFERENCES IN OPTIMAL GROWTH EQUATIONS FOR WHITE OAK IN THE INTERIOR HIGHLANDS

Don C. Bragg and James M. Guldin[1]

ABSTRACT.—Optimal growth equations are fundamental to many ecological simulators, but few have been critically examined. This paper reviews some of the behavior of the Potential Relative Increment (PRI) approach. Models for white oak were compared for Arkansas River Valley (ARV), Boston Mountains (BoM), Ouachita Mountains (OM), and Ozark Highlands (OH) ecological sections of the Interior Highlands. Noticeable divergence in equation shape was observed in the section and pooled models. PRI curves for the ARV and OM models predicted poor optimal growth, especially in the smallest size classes. The OH equation predicted high juvenile performance but limited large tree optima while the BoM model peaked at intermediate diameters. These distinctions may arise from differences in physiological potential between sections, or, more likely, from inadequate sample distributions. Our study supports pooling to improve optimal growth modeling if phenotypic conditions do not vary substantially.

Optimal growth equations are cornerstones of many ecological models (e.g., the gap models), but few existing designs have been thoroughly explored for their performance under different conditions. Without an understanding of how these equations will respond to different sampling conditions, size class distributions, or error structures, it is difficult to evaluate their quality. Recent critiques (e.g., Moore 1989, Zeide 1993, Vanclay 1994, Bragg 2001) have identified problems with some of the most commonly utilized potential growth models.

The Potential Relative Increment (PRI) approach to optimal growth modeling (Bragg 2001) has received some evaluation for behavior related to the derivation process (Bragg 2002), but there may be other fundamental issues associated with local or regional conditions that may affect this approach. Thus, using white oak (*Quercus alba* L.) as an example, this paper reviews additional PRI behavior as a function of ecological section and sample size.

STUDY AREA

The Interior Highlands of Arkansas, Missouri, and Oklahoma can be subdivided into four geographically and ecologically distinct sections (fig. 1). The northernmost section is the Ozark Highlands, which extends from south-central Missouri to northwestern Arkansas and northeastern Oklahoma. The Ozark Highlands grade into the Boston Mountains in eastern Oklahoma and west-central Arkansas. Uplifted, highly dissected limestone plateaus and deep, narrow river valleys dominate both of these sections. The Arkansas River Valley lies to the south of the Boston Mountains in central Arkansas and extends into eastern Oklahoma following the floodplain of the Arkansas River and the lower reaches of its tributaries. The southernmost section of the Interior Highlands is the Ouachita Mountains, a folded and faulted uplift of sedimentary origin characterized by east-west running ridges extending from central Arkansas to southeastern Oklahoma.

The Interior Highlands encompasses two primary ecological provinces (the Ozark Broadleaf Forest-Meadow and the Ouachita Mixed Forest-Meadow). Differences in parent materials, aspect, climate, and disturbance history have produced a mosaic of vegetation types across the study region. Presettlement vegetation

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differed markedly from one section to the next. The Ouachita Mountains were dominated by shortleaf pine (*Pinus echinata* Mill.), oak-pine, and oak-hickory communities, while the Arkansas River Valley ranged from oak-hickory or oak-pine forests on the well-drained sites to bottomland hardwoods. The Boston Mountains and Ozark Highlands were dominated by open woodlands interspersed with oak, shortleaf pine, and other hardwood cover types in sheltered coves and cedar glades on calcareous sites (Pell 1983). Many of these patterns have changed in the decades since Euroamerican settlement, with large areas cleared for agriculture, housing, industry, and mining or converted to other cover types through forest management.

PRI METHODOLOGY

The PRI methodology (Bragg 2001) involves the extraction of live tree records (those with positive growth) from the Eastwide Forest Inventory Data Base (EFIDB) (Hansen and others 1992), the selection of individuals growing at the highest rate within predetermined size classes (D_{MAX}), and the fitting of non-linear ordinary least squares regression equations to the final data using the following model form:

$$PRI = b_1 D_{MAX}^{b_2} b_3^{D_{MAX}}$$

where b_1, b_2, and b_3 are species-specific coefficients representing small tree growth potential, growth expansion factor, and the increment constraint factor, respectively. The software for this methodology is available from the primary author, and can be used for most species and state EFIDB inventories in the United States.

PRI equations for white oak in the Interior Highlands were derived to compare differences in behavior between the models for the Arkansas River Valley (ARV), Boston Mountains (BoM), Ouachita Mountains (OM), and Ozark Highlands (OH) ecological sections. White oak was chosen because of its abundance and commercial importance across the study region. Since the data in the EFIDB are presented by state or county and ecological sections were delineated without regard to these political boundaries, it was necessary to associate EFIDB plots with the appropriate ecological section. This was accomplished by translating plot locations to approximate section boundaries on a map and assigning plots to a section. For the few plots that were close to a boundary, current vegetation was used to help assign ecological section membership. Once sections were assigned, output files containing only the relevant plots were produced and then processed with the PRI methodology. A pooled model containing white oak data from across all sections was also fit for additional comparison.

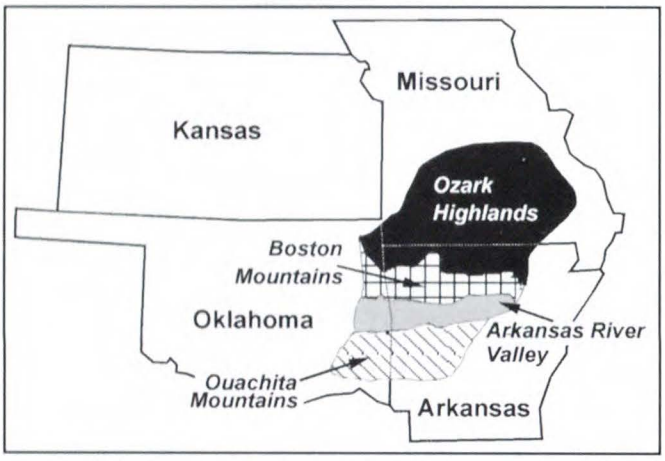

Figure 1.—*Map of the Interior Highlands, including section delineations.*

RESULTS

The ARV section contributed the fewest white oaks for PRI derivation (n = 384) compared to the other sections (fig. 2). All other sections yielded at least 1,600 individuals, with a total of 8,555 trees chosen across the Interior Highlands. Further consideration finds that not only are the samples unevenly distributed between the sections, their distributions are inconsistent within each section. Most small trees (< 15 cm diameter at breast height (d.b.h.)) and large

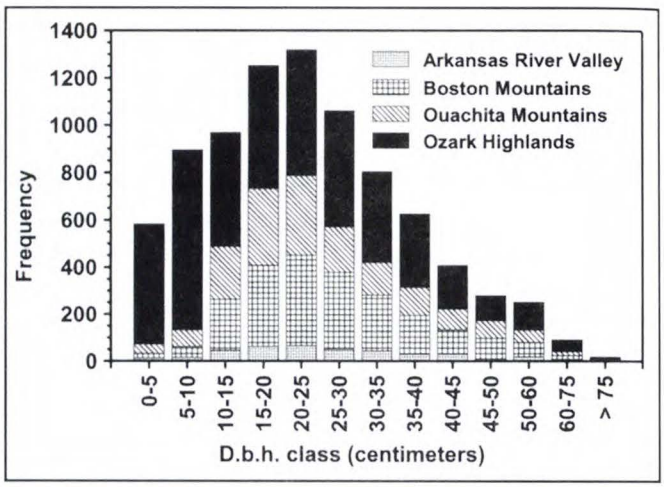

Figure 2.—*White oak sample distribution across the Interior Highlands by ecological section. Notice how trees taken from the Ozark Highlands sample dominated the smallest (< 10 cm d.b.h.) and largest (> 60 cm d.b.h.) classes, while the agriculture-dominated Arkansas River Valley contributed few individuals to any size class. Very few large trees were available, regardless of section.*

trees (> 50 cm d.b.h.) in the total sample were from the OH section, while the BoM and OM sections contributed primarily to intermediate diameters. The limited sample of the ARV section produced few representatives in any size class (especially the largest).

While there were few differences in minimum (all > 2.8 cm) and average white oak diameter (range = 21.5 to 26.8 cm) by section, considerable variation in maximum tree size was apparent (table 1). The biggest white oak by for the OM section was 74.2 cm d.b.h., compared to 85.3 cm for the ARV section, 93.7 cm in the BoM section, and 100.8 cm in the OH section.

Regression Models

Table 2 contains the fitted regression coefficients and measures of quality of model fit for each ecological section and the pooled model. The coefficients in table 2 can be roughly interpreted for their impacts on PRI models (Bragg 2001). High b_1 values suggest rapid growth in small diameter trees, which appears for both the OH and pooled models (fig. 3).

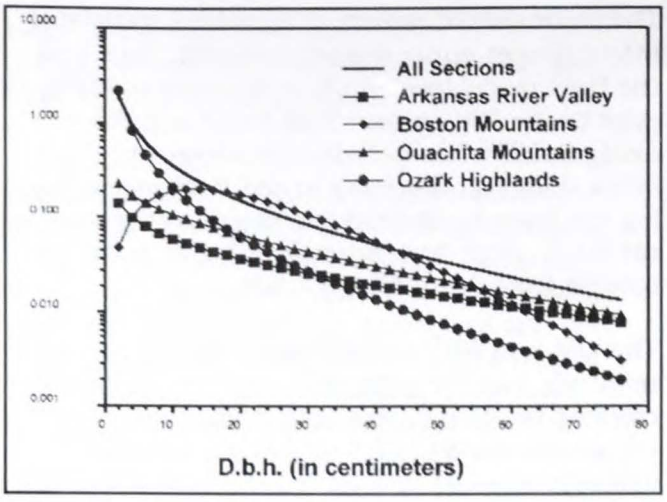

Figure 3.—*Potential Relative Increment (PRI) curves for white oak in the Interior Highlands ecoregion by ecological section and the pooled model. Note that the pooled model (solid line only) is a composite of all the section models, so its elevated performance reflects the fastest growing individual for the entire Interior Highlands.*

Table 1.—*Basic statistics on the sampled white oaks by ecological section*

Section (model abbreviation)	n	Minimum d.b.h.	Average d.b.h.	Maximum d.b.h.	Standard deviation
			cm		
Arkansas River Valley (ARV)	384	2.8	26.5	85.3	13.4
Boston Mountains (BoM)	2,012	3.0	26.8	93.7	11.7
Ouachita Mountains (OM)	1,678	2.8	25.1	74.2	12.4
Ozark Highlands (OH)	4,481	2.8	21.5	100.8	14.6
ALL SECTIONS (pooled)	8,555	2.8	23.7	100.8	13.7

Table 2.—*Regression coefficients by section*

Model	b_1	b_2	b_3	Final n[1]
OM	0.306556	-0.337021	0.973593	8
ARV	0.204727	-0.466773	0.983802	12
BoM	0.022778	1.381223	0.900305	10
OH	5.759565	-1.308236	0.969745	15
Pooled	3.817014	-0.935898	0.979100	10

[1] Final number of points used to fit optimal PRI curves.

The inconsistent nature of b_2 values translates into different curve shapes (table 2). Note how the BoM model (positive $b_2 > 1$) shows modality even in the PRI curves. This trend is further exaggerated when potential increment is calculated (fig. 4), resulting in the BoM model having the lowest potential increment at the smallest d.b.h. class but outperforming all other models from 17 to 42 cm d.b.h.

The OM and ARV models also displayed relatively high (though still negative) b_2 values, causing them to have slight modality in potential increment (fig. 4). The low negative b_2 values of the OH and pooled models limited the response of these equations, thus producing more of an exponential decline with increasing diameter. The PRI methodology requires that the b_3 be < 1, (b_3 values almost always exceed 0.90), since this parameter constrains diameter increment with increasing tree size. For example, the BoM model had a noticeably lower b_3 value, and displayed a greater reduction after its peak was reached.

Noticeable divergence in curve shape was observed between individual sections. Two of the models (ARV and OM) predicted poor optimal growth performance across the range of diameters considered, especially for the smallest size classes (fig. 4). This resulted in these models predicting annual optimal white oak increment of 0.5 to 1.0 cm, which is considerably lower performance than would be expected.

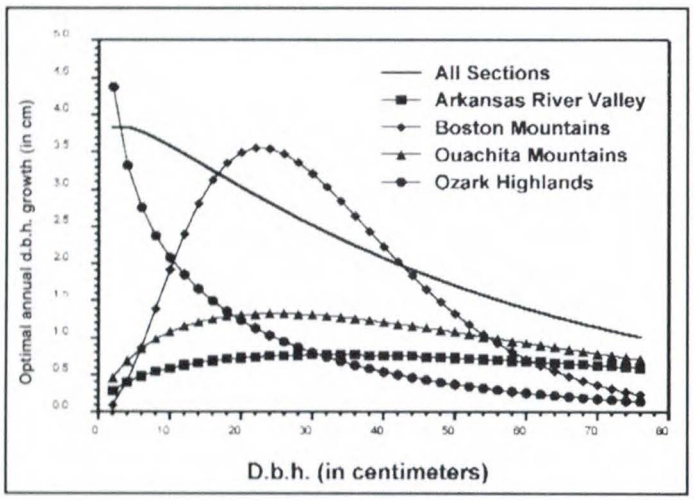

Figure 4.—*Multiplying the curves in figure 3 by their current d.b.h. yields predicted optimal annual d.b.h. growth for Interior Highland white oak. The pooled model, while generally outperforming other section models for optimal increment, was not always the highest performer.*

The OH model predicted high juvenile optimal growth (over 4 cm annually at the smallest diameters) but limited large tree performance (falling to the smallest potential increment of all models for trees 30 cm d.b.h. and larger). The BoM model peaked at intermediate sizes, with the lowest potential increment at the smallest diameters and the highest of all models from 17 to 42 cm d.b.h., followed by a decline to slow growth in big trees (fig. 4). The pooled model generally balanced all performance trends by utilizing the fastest growing individuals regardless of section. Thus, the local peaks experienced in OH and BoM models are attenuated by limited performance of other parts of the sample range.

DISCUSSION

The dissimilarity in optimal growth models by section may arise from several possible factors. One hypothesis suggests that there are innate differences between the physiological optimal growth performance of white oak growing in the ecological sections. These differences may be related to geography and topography, which could influence PRI analysis. For example, the proportion of oak or oak-pine forest types varies from 90 percent in the OH and BoM sections to 70 percent in the ARV section and 50 percent in the OM section (Guldin and others 1999).

In the OM section, white oak is more likely to occur on mesic north slopes, and thus may be found disproportionately on sites with less severe soil moisture deficits and appreciably higher productivity. Similarly, site quality increases as one goes from north to south across the Interior Highlands; thus, the better sites found in the ARV and OM sections may support oaks with inherently better growth. Management history could affect white oak growth performance. White oak was extensively cut for lumber and cooperage around the turn of the 20th Century, and the contemporary Interior Highland forests are largely a reflection of the genotypes found after these stands were high-graded. If there are inconsistencies in the age and genetics of the residual white oak timber, this could translate into the differential PRI response of this species across the ecological sections.

Another explanation of the observed disparity in optimal growth performance may be found in the differences in sample distribution. The histogram of section representation by size class (fig. 2) shows that very few small or large white oaks were sampled outside of the OH section. Therefore, the OH model dominates the

extremes of growth performance. Similarly, most intermediate size classes were dominated by the performance of the BoM section. The differential performance by section may arise from these incomplete diameter distributions. Since portions of the Interior Highland landscape have experienced shifting management and disturbance regimes, especially during the last 150 years, it is not surprising that the forests of each ecoregion have reached different developmental stages and thus would contribute differently to a regional PRI model.

The scarcity of white oaks in the largest (> 60 cm d.b.h.) size classes is typical of the maturing forests of the region, but under the PRI methodology this also means that the limited data exceeding this threshold have considerable influence on the final optimal growth models. In the original exploration of the PRI methodology, Bragg (2001) suggested that while the magnitude of error appeared highest in the intermediate size classes, the greatest potential for uncertainty lay in the biggest size classes because so few observations are found in that range.

The current age of the forests of the Interior Highlands is an unavoidable limitation on extending regional PRI models to include white oaks beyond 100 cm d.b.h. Additional flexibility may be possible if white oak from other ecoregions are incorporated, given that the phenotypic similarity between ecoregions would be acceptable (obviously different conditions should not be included if model integrity and robustness are to be preserved).

As can be seen in the pooled model, aggregating similar ecological sections provides some benefits when developing PRI models. The larger sample size ensures that fewer errors resulting from undersampled diameter classes occur in addition to supplementing the measured growth response in all classes. In this particular study, a pooled model allowed for the extension of the maximum d.b.h. from 74.2 cm if only the OM section were sampled to 100.8 cm if all sections of the Interior Highlands were included.

Broadly derived PRI models also improve regional predictability by expanding the range of sampling conditions. If adequately sampled, local (e.g., section- or compartment-based) optimal growth models can prove more accurate, but may be too site specific for regional application (Bragg 2001, 2002). A pooled model makes it theoretically possible to simulate growth performance from a dry southern or western facing slope in the Ouachita Mountains to the moist terraces along the Arkansas River or the sheltered cove forests of the Ozark Highlands.

CONCLUSIONS

Marked differences arose in white oak PRI growth models when a large ecoregion (the Interior Highlands of Arkansas, Oklahoma, and Missouri) was subdivided into four broad sections based on obvious ecological and geographic features. While it is probable that at least some of this variation may arise from genetic controls on individual trees or via section-based differences in the developmental dynamics of white oak, most of the inconsistencies can be attributed to varied sample distributions among sections. Poor representation of some (or all) size classes contributed to the widely different response curves.

This study supported pooling of regional inventory data to improve optimal growth modeling if environmental and genetics do not differ dramatically. By supplementing size classes and ameliorating local peaks (or valleys) in response curves, regional PRI fitting allows for a model capable of application across a broader range of size classes and environmental conditions.

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PROJECTION MATRICES AS A FOREST MANAGEMENT TOOL: AN INVASIVE TREE CASE STUDY

Ian J. Renne, Benjamin F. Tracy, and Timothy P. Spira[1]

ABSTRACT.—Life history parameters of many forest-dwelling species are affected by native and non-native pests. In turn, these pests alter forest processes and cost the United States billions of dollars annually. Population projection matrices can aid ecologists and managers in evaluating the impact of pests on forest species as well as devising effective strategies for pest control. As a case study, we use stage-classified projection matrices to assess the annual growth of five Chinese tallow tree (*Sapium sebiferum* (L.) Roxb.) populations and to evaluate the population growth consequences of simulated mortality among different stage classes. Four of five populations had positive annual growth over a 3-year interval and the growth average of all population-by-year matrices was +12 ± 5 percent. The effectiveness of management strategies differed such that the same strategy applied to all populations often resulted in differential local control. However, two of the same strategies were most effective in every population. It is thus important to know the demography of pest populations when control efforts are considered. We conclude by discussing how these adaptable models can be applied to common management issues. Examples include what should be done when immediate intervention is needed but life histories are not known, and how to achieve control objectives by using multiple management strategies.

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SOIL STRUCTURE CHARACTERIZED USING COMPUTED TOMOGRAPHIC IMAGES

Zhanqi Cheng, Stephen H. Anderson, Clark J. Gantzer, and J.W. Van Sambeek[1]

ABSTRACT.—Fractal analysis of soil structure is a relatively new method for quantifying the effects of management systems on soil properties and quality. The objective of this work was to explore several methods of studying images to describe and quantify structure of soils under forest management. This research uses computed tomography and a topological method called Multiple Resolution Blankets (MRB) to quantify soil structure. Differences among the signatures of macro-pore scale soil density were explored in intact cores removed under forest canopies. The MRB topological signatures are better able to separate treatment differences than the box-counting method that has been used in the past. The use of these techniques to quantify the effects of forest management systems on soil structure is encouraged.

Recently fractals have been used to examine heterogeneity of several natural structures, such as sandstone and fracture surfaces. Fractal analysis is a technique to evaluate whether materials with irregular surfaces are self-similar for multiple scales. Non-fractal objects are not self-similar. Soil structure is composed of an irregular fabric. Packed soils are homogeneous and therefore might be non-fractal. Intact soils have heterogeneous surfaces and have the potential to be fractal. Management of soils may affect whether or not soil structure is fractal. Soil quality in terms of soil structure may be quantified by evaluating whether or not a soil is fractal.

There are different methods for measuring the fractal dimension *D*. Not all fractal methods explain soil structure equally well because of the limitations in image resolution, size, and the limitations of the specific fractal method. Gimènez (1995) and Ogawa (1998) used a box-counting method to analyze soil images. Both found that the fractal dimension *D* is highly dependent on the threshold value chosen when converting a gray scale image into a binary image. Ogawa (1998) suggested that the assumptions required for mathematical fractal objects were never perfectly met.

Obtaining a succinct representation of a natural heterogeneous object is challenging. A 3-D probability box-counting fractal analysis method has been used to study soil structure obtained from computed tomography (CT) images of soil density. This method avoids problems caused by conversion of gray scale images into binary images; however, the method is not sufficiently sensitive to separate different soil structures without errors (Zeng and others 1996).

Albregtsen and others (1992) compared three methods of 3-D fractal analysis to distinguish between normal and malignant cell structure. They found that the Multiple Resolution Blankets gave the best discrimination between the two classes of cells (Peleg and others 1984). Talibuddin and others (1994) also concluded that the performance of the MRB method was the most reliable and efficient method for fractal parameter estimation on small datasets. The MRB signature shows promise for distinguishing soil structure and has the potential for developing better parameters to distinguish and describe soil structure. Unlike other fractal methods, the MRB signatures can also provide additional information on soil pore and solid arrangements for soil structures that other fractal methods cannot. The structural information

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can be used to quantify both fractal and non-fractal objects. Therefore, the objective of this research was to explore several methods of studying images to describe and quantify soil structure.

MATERIALS AND METHODS

Soil cores were taken from the A horizon of a Menfro silt loam soil (fine-silty, mixed, mesic Typic Hapludalf). Four 76 mm long by 76 mm diameter intact soil cores were taken within selected 1 m^2 sites in both a forest and grass field. Four soil cores were uniformly packed with air-dried soil collected from a cultivated field passed through a 2-mm sieve but retained on a 1-mm sieve. Another four soil cores were uniformly packed with soil that passed a 1-mm sieve. The average bulk density of cores was 1.10 ± 0.07 Mg m^{-3} for forest soil, and 1.51 ± 0.03 Mg m^{-3} for grass soil and 1.40 ± 0.01 Mg m^{-3} for packed soil. Air-dried soil cores were CT scanned and then oven-dried to constant weight to allow determination of the bulk density.

Computed Tomography (CT)

The resolution of the CT scanner was 0.1- by 0.1- by 2 mm thick (voxel size- the three dimensional equivalent of a two dimensional pixel). The scanner units are in Hounsfield units. The numerical value in the Hounsfield unit depends on the attenuation coefficient of the material relative to the attenuation of water. Output of the Hounsfield unit can be converted to gray scale values to display as images.

Soil MRB Signatures

The MRB method is designed to allow the calculation of the gray-scale surface area (A) that is produced using a wide range of blanket sizes (k). The area of the pixel value surface measured at different blanket thickness ($k^*\varepsilon$, where ε is a constant, and k is the blanket integer variable from 1 to an upper limit) will decrease at larger blanket thickness values. The gray-scale surface area is measured by calculating the volume between upper and lower blankets divided by 2ε. The SIG_C was calculated by finding the slope of the best fit straight line through the three points: $\{[\log(k-1), \log(A(k-1))], [\log(k), \log(A(k))], [\log(k+1), \log(A(k+1))]\}$.

The upper signature identified as SIG_U is derived from the higher gray scale values using the maximum operator to detect the surface characteristic of the higher value pixels. Lower fractal signature identified as SIG_L, is derived from the lower values and uses the minimum operator to measure the lower value pixels (Peleg and others 1984). These signatures ranged from 2.00 to 3.00. They were evaluated to measure the differences in the structural arrangements of soil cores. SIG_C was used to detect the heterogeneity and hierarchical nature of soil structure. SIG_U reflects the shape and size of pores, and SIG_L reflects the aggregates shape and size. Following the guidelines suggested by Talibuddin and others (1994), we set the lower limit of k as 10 and the upper limit of k as 40.

The difference between the MRB signatures from soil CT images were calculated using the following equation:

$$Dist(i,j) = \sum_{k=1}^{n} (SIG_{C_i}(k) - SIG_{C_j}(k))^2 \log\left(\frac{k+\frac{1}{2}}{k-\frac{1}{2}}\right)$$

For blanket integers of k = 10, 11, ...40, the smaller the difference between two soil cores based on their MRB signatures, the more similar the structures are and the more likely they can be grouped together (Cheng 2001).

RESULTS AND DISCUSSIONS

MRB Signature (SIG_C)

Figure 1 plots the average SIG_C vs. the blanket integer k. The shape of the SIG_C vs. k function for the packed cores had more strongly sloping curves with slopes of -0.0170 and –0.0120 compared to intact soil cores in the blanket integer of range from k = 10 to 40.

The average SIG_C slope of packed soil cores is about four times that of the intact soil cores (table 1). The intact soil cores had less sloping curves with slopes of -0.0043 and –0.0027. Images that have a constant SIG_C over the range of blanket integers are fractal. The signatures of all four soils were not flat lines.

For values of k ranging from 10 to 40, the SIG_C vs. k for the intact forest and grass soil cores was nearly flat indicating their density surface appears to be fractal within the range of k = 10 to 40 (fig. 1). For non-fractal objects, the SIG_C vs. k is characterized by variable SIG_C values and is represented by a sloping curve or line. The SIG_C of the density surface for packed soil cores indicated sloping curves; this indicated they were not fractal.

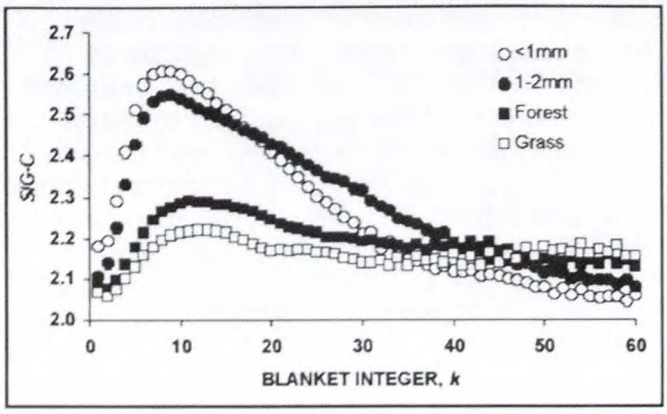

Figure 1.—*Average* SIG_C *signatures from MRB analysis used to detect the heterogeneity and hierarchical structure for intact forest and grass and packed soil cores.*

Figure 2 consists of two parts, a reproduced CT scan image, and a graph of the SIG_C vs. the blanket integer k. There were four replicated samples in each of the four groups. Variation among replicates in the < 1 mm packed core was the highest for the four treatments having a range in SIG_C of 0.17 at a blanket integer of 15 (fig. 2A). For the other three treatments, three of the four replicates had SIG_C values within 0.05 over the range of k. However, one sample replicate in each of these three treatments was more variable, having a variation value about 0.10 at a blanket integer of 15 (figs. 2B to 2D).

Soil aggregation may produce a fractal soil structure (Crawford and others 1993, Kozak and others 1996, Logsdon and others 1996). Soil densities measured by CT reflect the distribution of the soil aggregates. If aggregates are not fractal, their measured densities will not be fractal. Because packed cores have a smaller range in aggregate size relative to intact soil, these samples would be expected to be less fractal characteristics. This result agrees with the idea of Crawford and others (1993). They forward the idea that attempts to disintegrate aggregates sequentially through a clustered hierarchy, will not likely allow for a fractal model (packed soil). However, it is possible for intact soil to have a structure, which is fractal.

Lower Signature (SIG_L)

The average characteristic of the SIG_L was studied. Significant differences were found between graphs of the average SIG_L vs. blanket integer for soil cores from the four sources (fig. 3). The lower signature curve of packed soil looks like a bell, which increases sharply for short range distances ($k < 9$), then decreases more slowly at long range distance ($k > 9$).

The curve of the intact soil core, however, monotonically increases with a maximum value at the largest $k = 40$. The SIG_L value for the packed soil at high k was smaller. Peleg and others (1984) suggested that the SIG_L represents the shape and size of the aggregates and that the magnitude of the MRB signatures relates to the amount of details that are lost when the size of the measuring blanket passes k. Further, they suggest that high values of *MRB* signatures at small distance k result from significant high-frequency gray level variations while the high values for larger k result from significant low-frequency variations. Therefore, our results suggest that the aggregate sizes were smaller for the packed soil than for the intact forest or grass soil cores.

Upper Signature (SIG_U)

The shapes, but not the values of SIG_U, for the average upper signature SIG_U vs. blanket integer plots were similar for all four types of soil (fig. 4). The SIG_U reflects the shape and size of pores. The packed soil cores had high frequency variability across the lower range of blanket

Table 1.—*Average bulk density, multiple resolution blank signature values and fractal properties of soil from forest, grass, and packed cores*

Soil Core	Bulk Density	slope of SIG_C	SIG_C		SIG_U		SIG_L		
	Mgm^{-3}		MAX	MIN	MAX	MIN	MAX	MIN	
<1mm packed	1.40	-0.017	2.61	2.02	2.59	2.03	2.62	2.04	non-fractal
1-2mmpacked	1.40	-0.012	2.55	2.06	2.55	2.06	2.55	2.09	non-fractal
Forest	1.10	-0.004	2.29	2.08	2.41	2.00	2.30	2.04	fractal
Grass	1.51	-0.003	2.22	2.05	2.37	2.00	2.39	2.02	fractal

Note: SIG values range from 2.00 to 3.00.

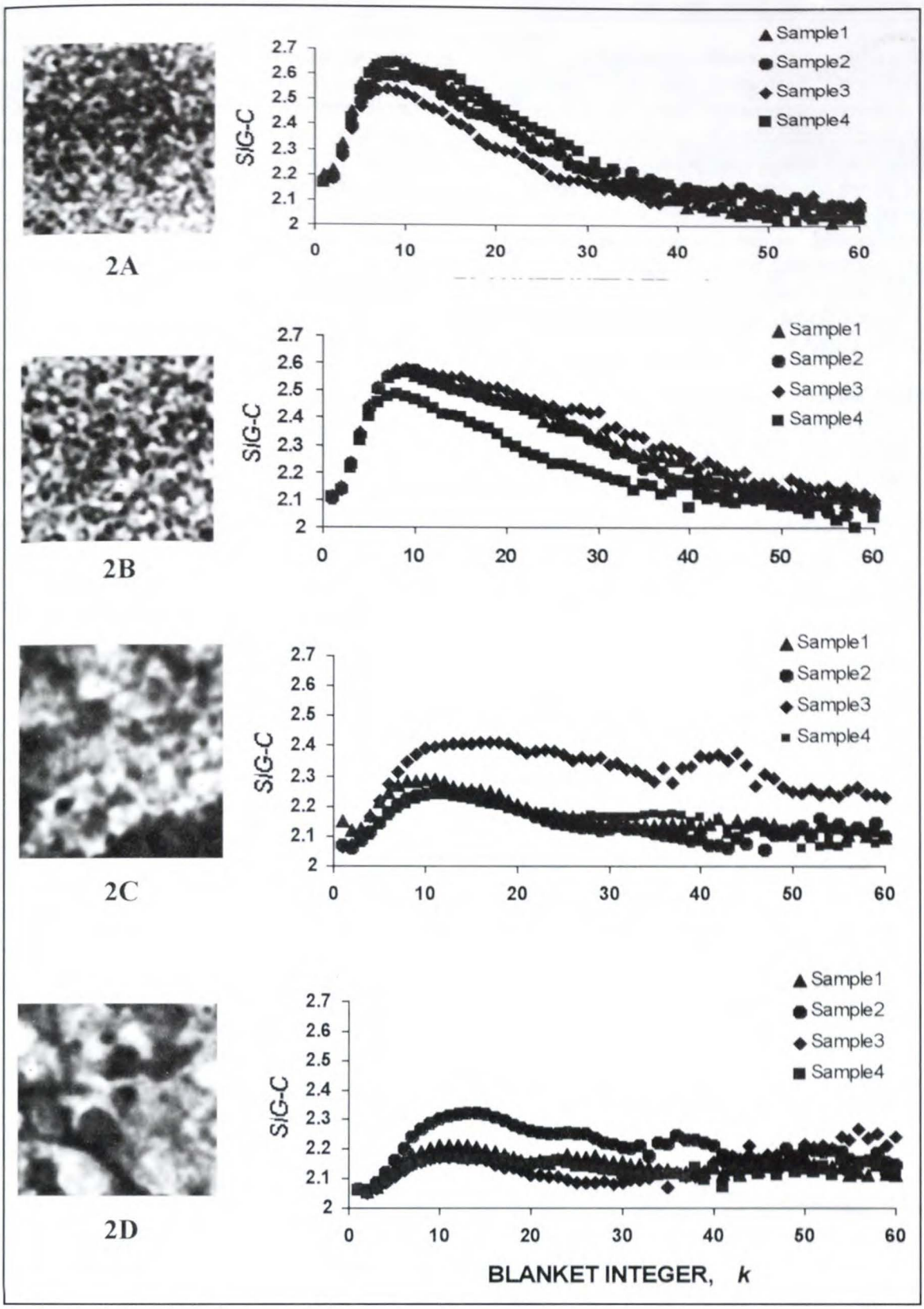

Figure 2.—*Average gray scale image and individual soil core* SIG_C *signature from MRB analysis of 256 by 256 pixel CT-images from packed soil cores (2A and 2B) and four intact forest (2C) and grass cores (2D).*

integer, indicating that the pore sizes in packed soil were smaller than pore sizes for intact soil.

Our results agree with results reported by Zeng and others (1996), indicating that the CT-measured soil density surfaces for the packed cores had peaks and valleys that are of high frequency. The intact soil cores had peaks and valleys that are of lower frequency. The average signature curve of grass and forest soil cores are almost overlapping, as are the signature curves for the two packed soil cores. These results suggest that the pore shape and size were different between classes (intact vs. disturbed soil) but similar within classes.

Soil Structure Difference Analysis

Calculated soil structure differences between the MRB signatures clearly separate the soil cores into groups for packed and intact soils (table 2). The only incorrectly classified image is the third replication for the forest soil cores, which fell into the packed 1 to 2 mm soil group. This misclassification may have been due to the high variability found in soils under forest management.

For each soil treatment, the three cores with the smallest soil structure difference occurred within the correct soil treatment only 50 percent of the time for < 1 mm soil cores, 58 percent of the time for 1 to 2 mm soil cores, 58 percent of the time for forest soil cores, and 42 percent of the time for grass soil cores. Therefore, this method is not suited for analyzing soil structure classification using CT images for similar classed soil structure. However, it still does a good job of separation between classes (disturbed vs. intact).

MRB vs. Box-Counting Method

Zeng and others (1996) used a box-counting method to estimate the fractal dimension (D) and lacunarity $C(L)$ of the same images used in this study. They also used these parameters to evaluate soil density distribution of different soil groups. The box-counting method indicated that cores of all soil groups were fractal, when in fact, only the intact cores were found fractal with the MRB method. Plotting the fractal dimension against lacunarity $C(L)$ using the box-counting method did not resolve soil cores into unique groupings (fig. 5).

Plotting the MRB signature values for SIG_C when $k = 10$ against the slope (change in SIG_C divided by change in k) clearly separated packed soil from intact soil (fig. 6). Good separation of intact cores from forest and grass soils is also achieved. This figure provides for an approximate graphical method of determining if the groups are distinct. As can be seen, the forest and grass soil and the packed soil are completely separated. Comparison with the classification using lacunarity and fractal D shows that the box-counting method is not as powerful as the MRB method (figs. 5 and 6).

CONCLUSIONS

Soil density of cores packed by single sized aggregates was found to be non-fractal, while intact soils show a fractal density distribution. The differences between soil structures with different treatments are caused by the aggregation size and shape differences. The structural difference measurement allows us to group soil samples that are similar. The MRB signatures are better able to separate treatment differences

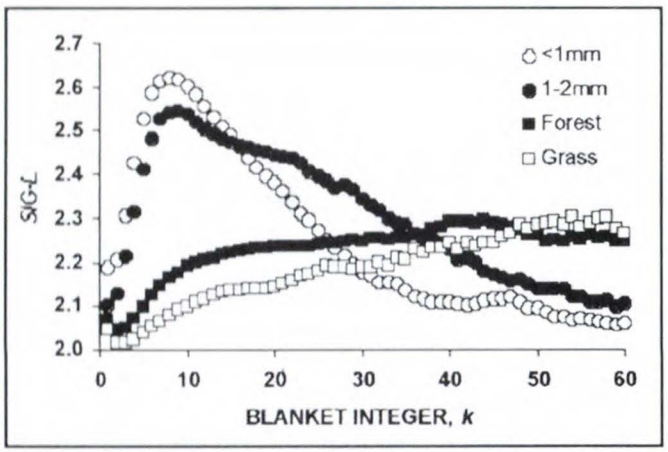

Figure 3.—*Average* SIG_L *signatures from MRB analysis used to describe the shape and size of aggregates for intact forest and grass and packed soil cores.*

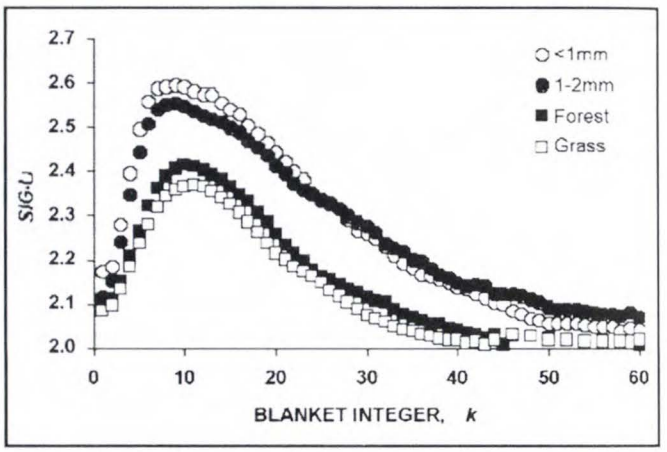

Figure 4.—*Average* SIG_U *signatures from MRB analysis used to describe the shape and size of pores for intact forest and grass and packed soil cores.*

Table 2.—*Calculated* MRB *soil structure distance between each soil core using* SIG_C *for k = 10 to 40. Values for the three cores most similar to the sample core are underlined within each row.*

Treatment	Smpl.	TREATMENT: <1 mm Packed No. 1	<1 mm Packed No. 2	<1 mm Packed No. 3	<1 mm Packed No. 4	1 - 2 mm Packed No. 1	1 - 2 mm Packed No. 2	1 - 2 mm Packed No. 3	1 - 2 mm Packed No. 4	Forest Intact Cores No. 1	Forest Intact Cores No. 2	Forest Intact Cores No. 3	Forest Intact Cores No. 4	Grassland Intact Cores No. 1	Grassland Intact Cores No. 2	Grassland Intact Cores No. 3	Grassland Intact Cores No. 4
<1mm	1	0.000	0.004	0.025	0.002	0.010	0.011	0.014	0.042	0.176	0.224	0.091	0.238	0.257	0.160	0.316	0.300
	2	0.004	0.000	0.010	0.007	0.005	0.005	0.011	0.021	0.128	0.170	0.060	0.182	0.198	0.114	0.250	0.236
	3	0.025	0.010	0.000	0.033	0.017	0.018	0.032	0.004	0.074	0.107	0.042	0.118	0.130	0.070	0.170	0.160
	4	0.002	0.007	0.033	0.000	0.009	0.009	0.009	0.049	0.188	0.237	0.090	0.250	0.269	0.167	0.332	0.314
1 - 2mm	1	0.010	0.005	0.017	0.009	0.000	0.000	0.003	0.023	0.124	0.164	0.044	0.173	0.189	0.104	0.244	0.226
	2	0.011	0.005	0.018	0.009	0.000	0.000	0.002	0.024	0.126	0.166	0.044	0.175	0.191	0.105	0.246	0.228
	3	0.014	0.011	0.032	0.009	0.003	0.002	0.000	0.038	0.151	0.194	0.053	0.202	0.220	0.125	0.281	0.260
	4	0.042	0.021	0.004	0.049	0.023	0.024	0.038	0.000	0.047	0.074	0.024	0.082	0.093	0.043	0.128	0.119
Forest	1	0.176	0.128	0.074	0.188	0.124	0.126	0.151	0.047	0.000	0.003	0.037	0.005	0.008	0.005	0.020	0.017
	2	0.224	0.170	0.107	0.237	0.164	0.166	0.194	0.074	0.003	0.000	0.054	0.001	0.002	0.009	0.008	0.006
	3	0.091	0.060	0.042	0.090	0.044	0.044	0.053	0.024	0.037	0.054	0.000	0.056	0.066	0.019	0.102	0.088
	4	0.238	0.182	0.118	0.250	0.173	0.175	0.202	0.082	0.005	0.001	0.056	0.000	0.001	0.011	0.007	0.004
Grass	1	0.257	0.198	0.130	0.269	0.189	0.191	0.220	0.093	0.008	0.002	0.066	0.001	0.000	0.015	0.004	0.002
	2	0.160	0.114	0.070	0.167	0.104	0.105	0.125	0.043	0.005	0.009	0.019	0.011	0.015	0.000	0.034	0.026
	3	0.316	0.250	0.170	0.332	0.244	0.246	0.281	0.128	0.020	0.008	0.102	0.007	0.004	0.034	0.000	0.001
	4	0.300	0.236	0.160	0.314	0.226	0.228	0.260	0.119	0.017	0.006	0.088	0.004	0.002	0.026	0.001	0.000

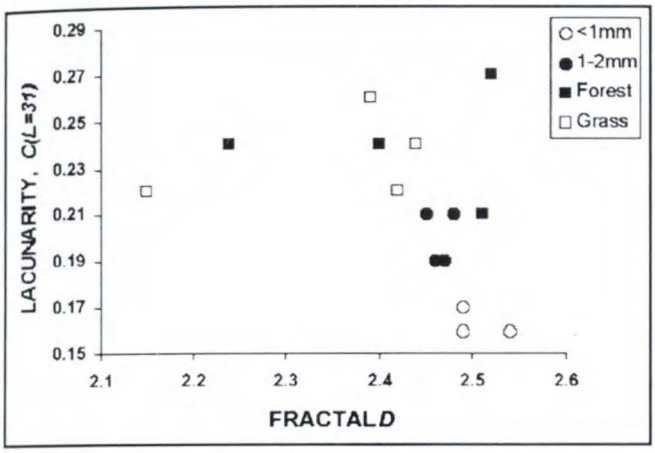

Figure 5.— *Fractal lacunarity, C(L), vs. fractal dimension,* D, *from intact forest and grass and packed soil cores (after Zeng 1996).*

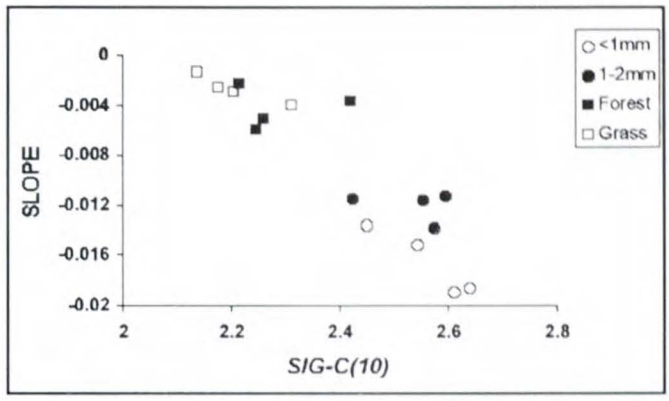

Figure 6.— *Slope of* SIG_C *for blanket integer ranging from 10 to 40 vs.* SIG_C *(10) value by* MRB *method from four groups of soil samples.*

than the box-counting method using graphical techniques. The MRB method showed greater discrimination between intact and sieved soil compared to the box-counting method.

The MRB parameters also provide additional information about soil structure. These parameters are related to size and distribution of aggregates and pores. This method can be used for both fractal and non-fractal images. Two different measures can be obtained, the upper and lower signatures, which provide measurement information on both the arrangement and the distribution of pores and solids. These measures appear to provide more information than the fractal *D* and lacunarity together (Albregtsen and others 1992, Talibuddin and others 1994, Zeng and others 1996). Future work is suggested for using these techniques in comparing the effects of forest management systems on soil structure.

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ASSESSING WEST VIRGINIA NIPF OWNER PREFERRED FOREST MANAGEMENT ASSISTANCE TOPICS AND DELIVERY METHODS

Daniel J. Magill, Rory F. Fraser, and David W. McGill[1]

ABSTRACT.—Four hundred and fourteen non-industrial private forest (NIPF) owners in West Virginia responded to a mail survey questionnaire assessing their forest management assistance topics and delivery methods of interest. Logistic regression was used to analyze 39 independent variables in relation to the dependent variables of wanting a specific topic of forestry assistance or not. Ownership of property for investment, cultivation of wildlife food crops, receiving assistance from the West Virginia State Division of Forestry, and ownership tenure were recurrent significant variables characterizing landowners wanting a specific assistance topic. These results can be used to develop forestry assistance programs that reflect landowner objectives and good forest management.

Forests cover nearly 80 percent of the West Virginia land base. The resources derived from those forests are vital for the economic, social, and ecological sustainability of the State and its residents. In 1995, forest-based activities such as timber harvesting, saw milling, and wood products manufacturing contributed $3.2 billion to the West Virginia economy. Nature based recreation and tourism contributed $1.5 billion to the States economic output in 1996 (Magill 2000). Individual benefits such as privacy of residence, freedom of ownership, and a sense of place are also derived from forest lands like those of West Virginia. Other social benefits provided by such forest land include un-priced forest recreational and subsistence uses, aesthetics, clean water, and fresh air (Flora 1992). More important, the benefits provided by forest resources are renewable over the long-term, if they are properly managed.

White (1993) reported that 83 percent of the forest land in West Virginia belonged to private landowners. According to Birch (1996), 76 percent of the 10,745,000 privately owned forest acres in the State belonged to non-industrial private forest (NIPF) owners. Given the large proportion of NIPF ownership (> 250,000) in the State, the concern is that less than 15 percent of these landowners are known to have received assistance in the last 10 years (Fraser and Magill 2000). Further heightening concerns to provide sufficient forestry assistance is the continued harvest of timber over the last decade using diameter limit cutting, especially on NIPF properties in West Virginia (Fajvan and others 1998). The problem is that diameter limit cutting, especially when repeated, can degrade the economic, social and ecological quality of these privately owned woodland resources.

The primary objective of this study was to determine what forest management topics NIPF landowners in West Virginia prefer and whether or not those preferences are influenced by their socioeconomic characteristics, their reasons for owning forest property as well as management activities conducted on their property, and sources of assistance received. A secondary objective was to determine whether landowners preferred one educational assistance delivery method to another.

METHODS

NIPF landowner information for the study was obtained during the winter of 1999-2000 using a stratified random sample (by region, forest cover, and acreage) and a seven-page mail survey questionnaire. Question categories included property size and ownership, forest land use

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Table 1.—*Independent landowner charactistics analyzed*

Demographics	Property uses	Forest activity	Agencies
Region	Residence	Harvest timber	WV University Forestry
Land size	Investment	Plant trees	WV University Extension
Tenure	Wildlife habitat	Thin trees	WV State Forestry
Income	Hunt or Fish	Cut vines	
Landowner age	Watershed	Visual resource	
Education level	Visual beauty	Build trails	
	Hike or Bike	Build roads	
	Produce timber	Wildlife food	
	Christmas trees	Soil resource	
	Fruit orchard	Water resource	
	Ginseng etc.	Ginseng, etc	
	Graze livestock	Survey property	
	Firewood /Posts	Forest protection	
	Maple syrup	Lease property	
	Other	Other (fencing)	

and management, assistance received, socioeconomic information, and assistance preferred. Questionnaires were sent to 1,080 forest owners using multiple mailings based on the Dillman Method: first mailing, follow-up postcard, and then second mailing to increase response rate (Dillman 1978).

Logistic regression (logit) models were used to narrow down and identify the most significant group of explanatory landowner attributes that differentiate those interested in a specific assistance topic. The dependent variables examined for analysis were chosen from the assistance topics included in the survey questionnaire. Nine of the fifteen assistance topics surveyed were analyzed using logistic regression and were chosen based on the number (≥ 40 percent of 414) of owners (≥ 166) requesting each topic of assistance. The number of assistance topics was narrowed to nine for logistic analysis in order to concentrate on the most requested types of management aid.

Logistic regression was performed separately for each of the topics analyzed using 39 independent landowner variables and the dependent variable (binary Yes = 1 or No = 0) of wanting a certain topic or not. The explanatory (independent) variables included 6 demographics, 15 property-uses, 15 management activities, and 3 agencies (table 1).

The significant independent variables were then retested against each other separately by topic to provide the most parsimonious set of explanatory factors that describe landowners wanting a certain topic of assistance. All statistical significance levels were set at alpha ≤ 0.10. Backward elimination was used to select independent variables by significance of p-values (≤ 0.10) in the model building process. Explanatory (independent) variable p-values were used to test the null hypothesis (H_o: $b_i = 0$) that selection of a specific forestry assistance topic was independent of all landowner characteristics. Descriptive statistics were computed to highlight differences in educational delivery mechanisms preferred by landowners wanting a specific assistance topic.

RESULTS

The sample response of 414 landowners proved to be representative across the State, which was demonstrated by the lack of significance ($Chi^2 = 1.91$, $p = 0.72$) between the number who responded and the number expected to respond according to the sample stratification.

The most sought after assistance topic was forest damage prevention (e.g., by fire, insects, disease, deer, etc.) followed by water resources, silviculture, wildlife management, landowner liability, property rights and taxes, soil resources, aesthetic improvement, dendrology and silvics, valuation of timber or forest land (fig. 1).

Chi^2 values for all nine logistic regression models did not change significantly with the reduction in explanatory independent variables, making the models with less significant (independent) factors parsimonious for identifying respondents wanting a specific topic. This means that fewer characteristics (average of 5) were needed at the alpha ≤ 0.10-level to identify landowners interested in each assistance topic.

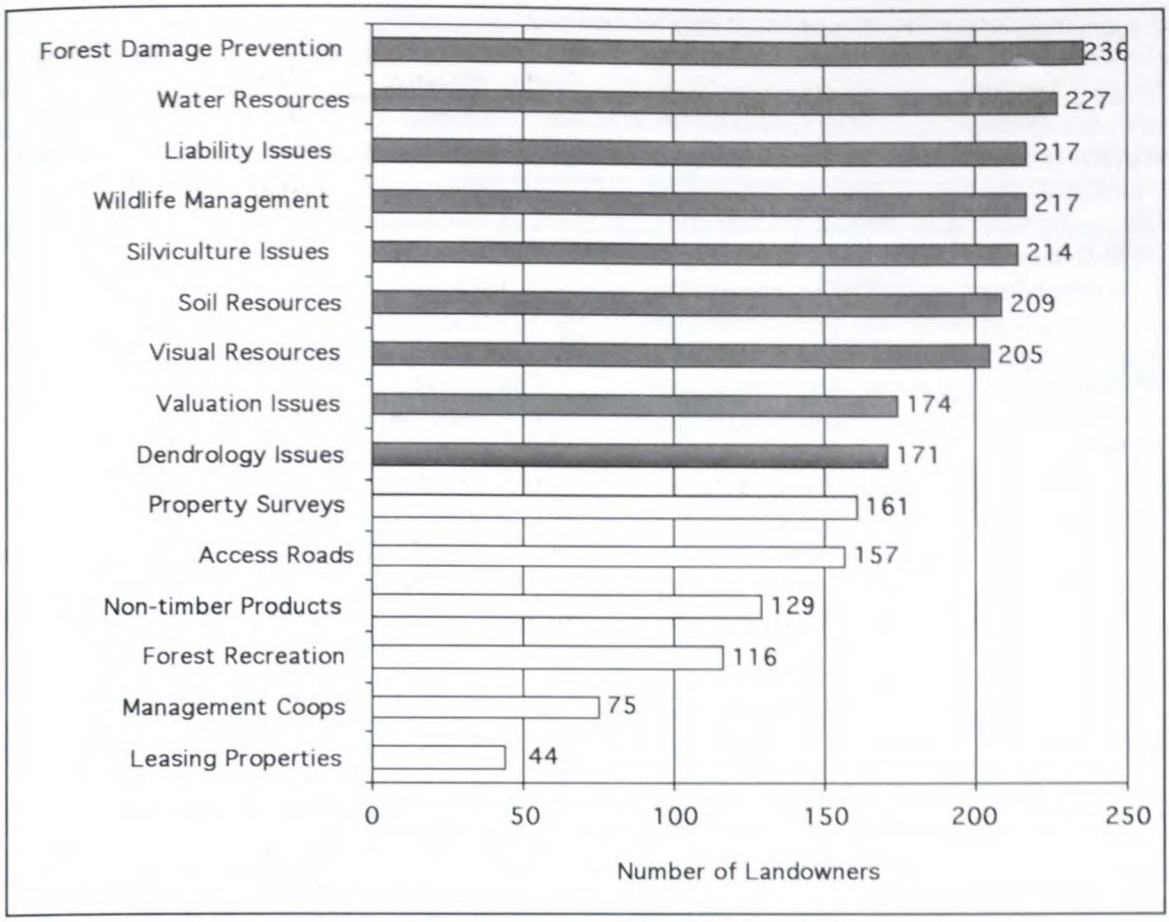

Figure 1.—*Assistance topics preferred by respondents (gray bars = topics analyzed).*

The results from the parsimonious models formed the basis for statistical conclusion and discussion in this paper.

The models, on average, accurately identified 70 percent interested and 64 percent of those not interested, respectively, in the topics. The odds-ratio on average was 5:1 meaning that landowners were 5 times more likely on average to request one of these nine topics when characterized by the significant explanatory variables (table 2). Negative signs accompanying p-values indicate a negative relationship between that variable and the topic of interest. In this analysis the negative relationships mean a smaller land-size, less tenure, and younger landowner age.

Logistic Regression Results

Results from logistic regression analysis models revealed similarities and differences between landowners regarding their preference for different assistance topics. The models linked these preferences with landowner reasons for owning forest property, forestry practices conducted, assisting agencies, and socioeconomic characteristics. Ownership of property for investment, cultivation of wildlife food crops, length of ownership, and receiving assistance from the West Virginia State Division of Forestry were recurrent significant factors characterizing landowners interested in the nine topics analyzed (table 2).

Investment was an inherent attribute of forest ownership for landowners interested in forest damage prevention, wildlife management, silviculture, forest resource valuation, dendrology, and liability issues. NIPF owners in Louisiana also indicated investment as an important ownership objective (Lorenzo and Beard 1996). Respondents requesting assistance for wildlife and visual resource management, soil fertility, and erosion issues as well as for plant identification were more likely to plant wildlife food crops. Landowners requesting wildlife management assistance also tend to own their forest property for wildlife habitat and a place to live. Studies in Kentucky (Gracey and Pelkki 1996), Indiana (Mills and others 1996), and Louisiana (Lorenzo and Beard 1996) likewise indicated that NIPF owners in those states tend to have wildlife ownership and/or management objectives.

Table 2.—*Logistic regression significant p-values and odds ratio by assistance topic*

Characteristics	Forest damage prevent	Water resource	Liability issues	Wildlife Manage.	Silvi-culture issues	Soil resource	Visual issues	Valu-ation issues	Dendro-logical issues
Odds ratio	4 to 1	4 to 1	4 to 1	6 to 1	6 to 1	3 to 1	3 to 1	5 to 1	4 to 1
Demographics									
Land size		0.04 (-)							
Tenure			<0.001 (-)	<0.001 (-)					<0.001 (-)
Income		<0.001		0.02					
Landowner age						<0.001 (-)			
Property uses									
Residence				0.02					
Investment	<0.001		0.02	0.02	0.01			0.01	<0.001
Wildlife habitat				<0.001					
Visual beauty					<0.001		<0.001		
Produce timber	<0.001							<0.001	
Ginseng etc.			0.04						
Graze livestock		<0.001				<0.001			
Forest activity									
Cut vines								0.04	
Build trails				<0.001					
Wildlife food				0.01		0.02	0.09		0.01
Ginseng, etc.		0.01			0.02				
Forest protection	<0.001		<0.001				0.03		
Other (fencing)					<0.001	0.03		<0.001	
Agencies									
WV State Forestry	<0.001	<0.001	<0.001		<0.001	0.01	<0.001	<0.001	

Owning forest property for a shorter period of time (1 to 10 years) was characteristic of landowners wanting aid to deal with liability, wildlife, and plant identification issues. Landowners earning higher annual incomes (> $40,000) were more inclined to request wildlife management and water resource assistance (table 2). This was comparable to an Indiana study in which NIPF owners participating in government forestry assistance programs were found to have higher incomes and own forest property for less years than did non-participants (Mills and others 1996). Receiving assistance from the West Virginia State Division of Forestry was an attribute characteristic of respondents interested in seven of the nine topics tested. NIPF owners in Kentucky also indicated state forestry agencies as a more relevant means to obtain management assistance (Gracey and Pelkki 1996).

Production of timber and visual forest enjoyment were the second most significant forest ownership reasons for landowners interested in the topics analyzed. Timber production was a differentiating ownership characteristic of landowners requesting forest damage prevention and valuation assistance. Cultivation of timber resources was likewise a distinct property use for NIPF owners belonging to the Kentucky Woodland Owners Association (Gracey and Pelkki 1996). Maintaining visual resources was a distinct ownership objective for West Virginia respondents interested in visual resources and silvicultural issues (table 2). NIPF owner's interested in silviculture aid were more inclined to grow and harvest non-timber products (ginseng, etc.), as were those with water resource and liability concerns.

Implementation of forest damage prevention and fence-building practices followed wildlife food plantings as the most common significant management activities conducted by landowners requesting the nine topics (table 2). Protecting forest resources was a distinguishing management practice implemented by landowners having forest damage prevention, visual resource, and ownership liability concerns. A

study in Indiana concluded that NIPF owners participating in government programs tend to request forest damage prevention assistance (Mills and others 1996). Constructing fence to protect forest resources was a significant management activity for landowners interested in silviculture practices, soil resources, and valuation issues.

Livestock pasturage was a distinctive property use for landowners interested in both water and soil resource issues (table 2). A Kentucky study likewise revealed farming to be an important ownership reason for the NIPF population in that state (Gracey and Pelkki 1996). Building access trails and cutting vines were significant management activities for only the landowners wanting wildlife management and valuation assistance, respectively.

Preferred Educational Delivery

With regards to delivery mechanisms, workshops were most requested by landowners (63 percent) wanting to address liability concerns followed by those wanting to address visual resource management, plant identification, wildlife management, timber and forest land valuation, forest damage prevention, and silviculture issues (table 3). Financial management aid was the most sought after by respondents wanting water (68 percent) and soil (65 percent) management assistance followed by landowners seeking assistance for wildlife management, silviculture, and forest damage prevention. Technical assistance services were of the most interest for landowners wanting to obtain water management (74 percent) and forest damage prevention (70 percent) aid followed by respondents wanting aid to address soil resource, silviculture, wildlife management, valuation, and liability concerns.

DISCUSSION AND IMPLICATIONS

The response to this survey by NIPF landowners was enthusiastic and revealed interest in a wide range of forestry assistance topics. Results indicate that this population of NIPF landowners interested in obtaining help is varied, because selection of a topic in this analysis was more dependent on reasons of forest ownership and management activities conducted than on the socioeconomic characteristics of the landowner. The landowners that participated in this survey see their forest property as an investment for varied reasons.

There is a high level of interest in wildlife management, which concurs with the results of NIPF studies in other states. Timber production and visual forest enjoyment, which were revealed as common attributes of ownership in this study, solidify landowner interest in varied forest resource use and management. Implementation of forest protection practices were also among the most influential management activities indicated in relation to selecting a topic of assistance and further illustrate investment as an important reason of forest ownership.

Forest damage prevention was the most requested assistance topic, which further substantiates the concern of this landowner group to protect their forest investment. Methods of educational and assistance delivery preferred by respondents were likewise diverse, but the highest proportion of requests were for technical aid (table 2).

Previously obtaining assistance from the West Virginia State Division of Forestry was likewise an influencing factor for a respondent selecting one of the nine topics included in this survey. Perhaps landowners used this state agency for

Table 3.—*Preferred delivery mechanism percentages by assistance topic*

		Preferred delivery mechanism		
Assistance topics	**Requests**	**Workshop**	**Financial**	**Technical**
Forest damage prevention	236	50%	51%	70%
Water resources	227	44%	68%	74%
Liability issues	217	63%	27%	59%
Wildlife management	217	54%	57%	67%
Silviculture issues	214	49%	57%	70%
Soil resources	209	44%	65%	71%
Visual issues	205	60%	47%	61%
Valuation issues	174	52%	33%	79%
Dendro issues	171	59%	51%	65%

previously obtaining assistance, because it is more visible and available. State forestry agencies were also indicated by a Kentucky study of NIPF owners as a more relevant source of assistance. This same study confirmed that landowners belonging to the Kentucky Woodland Owners Association were more actively involved in managing their forest properties (Gracey and Pelkki 1996). Getting more West Virginia landowners to join forestry organizations may improve their participation in forestry assistance programs, which in turn would help landowners to better understand and actively implement good management practices.

Forestry assistance and information programs in West Virginia should focus on enabling landowners to meet their ownership and management objectives. These programs should additionally focus on fostering more landowners who are better informed about and who implement good forest management practices. Assistance topics provided should cover a diverse array of forest management activities in order to involve a wider landowner audience. Utilizing a variety of methods and programs for engaging all NIPF owners could further result in better management of forest resources in West Virginia. In addition to involving more landowners in organizations like the Woodland Owners Association, these program focuses used in combination should likewise help to improve landowner participation (table 2). The computer internet is another mode of information delivery not analyzed in this study that could prove highly useful for increasing forest landowner participation in management and assistance programs.

Kluender and Walkingstick (2000) suggest that concentrating landowner education on a variety of ownership and management objectives would foster a better relationship between resource professionals, forest industry, and environmentally concerned landowners while likewise making forest owners more likely to manage their resources. Results from this survey could be used to develop forestry assistance and information programs in West Virginia that reflect the objectives of these landowners as well as good forest management.

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DIVERSIFYING FARM INCOME THROUGH WOODLOT MANAGEMENT: CHARACTERISTICS OF FARMER-OWNED NIPF LAND IN ILLINOIS

Eric M. White and Andrew D. Carver[1]

ABSTRACT.— The lessening dependence on public lands for the supply of timber and a shift in the focus of timber procurement from the western U.S. to the South and Midwest, coupled with the indications of increasing lumber production rates in the Midwest, may signal a significant opportunity for NIPF landowners in Illinois. The objective of this study is to determine the forest characteristics and spatial patterns of farmer-owned NIPF timberland for counties in Illinois in an effort to identify those areas in Illinois where farmers have the greatest opportunity to capitalize on existing NIPF acreage. The study is the first step in quantifying the opportunities for increased timber harvesting from this NIPF ownership group within Illinois. Counties located in western and southern Illinois may have the greatest potential for increased timber harvesting on farmer-owned NIPF based upon forest acreage, growing-stock volume, site classification, and forest tract size.

As a result of litigation and changes in public sentiment, timber harvesting has declined on publicly owned land throughout the United States and, in particular, the West. This reduction in the available supply of timber resources from publicly owned lands have shifted the focus of timber supply to nonindustrial private forest (NIPF) lands. Increasing demand for timber resources from NIPF land may suggest that timber supplies in the Midwest will fall under increased procurement pressure from forest products industry (Callahan 1980, Wiedenbeck and Araman 1993). Recently, an increase in the production of hardwood lumber from states located in the lower Midwest has been documented by Luppold and Dempsey (1994).

The lessening dependence on public lands for the supply of timber and a shift in the focus of timber procurement from the western U.S. to the Midwest and South, coupled with indications of increasing hardwood lumber production in the Midwest, may signal a significant income opportunity for NIPF landowners in Illinois. Those NIPF landowner groups with propensity to harvest timber will have the greatest chance of benefiting from increased demand on the NIPF timber supply. Within Illinois, farmer-owned NIPF timber acreage has the greatest rate of harvest of all NIPF groups in the State (Birch 1996). In general then, farmers in Illinois, who own existing NIPF acreage may be in a good position to benefit from increasing demand for timber.

The degree to which Illinois farmers will be able to capitalize on increased demand for timber will be determined by the degree to which small NIPF tracts can satisfy the biological and economic constraints of harvesting operations. Loggers and timber buyers in Massachusetts who purchased timber from small NIPF tracts identified growing-stock volume and timber quality as the biological characteristics of greatest concern when deciding to bid on a potential harvesting operation (Kittredge and others 1996). Additionally, timber tract sizes must be of sufficient size to satisfy the economic constraints of operation costs.

The objective of this study is to determine the forest characteristics and spatial patterns of

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farmer-owned NIPF timberland for counties in Illinois in an effort to identify those areas in Illinois where farmers have the greatest opportunity to capitalize on existing NIPF acreage. This study is the first step in quantifying the opportunities for increased timber harvesting from the farmer-owned NIPF within Illinois.

DIVERSIFIED FARM INCOME

Agriculture in Illinois is highly concentrated in soybean and corn production. In 1997, 90 percent (20 million acres) of the State's cropland was dedicated to corn and soybean crops. Concordantly, 75 percent of the farm income was manifested from the sale of corn and soybean production to the market. Such figures suggest that Illinois has significant opportunity to diversify agriculture production, and thereby income sources, through the planting of a greater variety of crops and the inclusion of alternative production in farm strategies. The harvesting of timber from existing farmer-owned NIPF land is one means of diversifying Illinois' monotonic agriculture production using an alternate production strategy.

Income generated from forestry activities has been shown to positively impact gross farm income within the Midwest and Illinois. A study of corn growers in the 16 major corn-producing states found that 5.5 percent received some income from forestry operations in 1996 (Kelson 1999). While the number of farmers taking advantage of forest income is small, the income created can be substantive.

Income from forest related activities created an average of $10,378 in gross income (9.1 percent of total farm gross income) for those farms that included forestry activities in their production strategies in 1996 (Kelson 1999). Total gross income created through forestry activities for all corn producers in the study was $207 million in 1996 (Kelson 1999). Farmers in Illinois created over $3 million in income from the sale of forest products (excluding Christmas tree and maple syrup sales) in 1997 according to the National Agriculture Statistics Service (NASS). Statewide in 1997, 719 farms in Illinois created income from the sale of forest products (NASS 1997).

Small to moderate-sized farms struggling to remain successful competing in concentrated crop production markets may do well to allocate more land (and resources) to forest production. This may be particularly applicable to Illinois where a steady decline in the number of farms in operation has occurred over the last 3 decades. Farms most likely to fail in Illinois are those owned by individuals or partnerships and those that are less than 500 acres in size. This is opposite to the growth trends found in farms owned by corporations and of farms greater than 500 acres in size. Additional income created through forestry activity may be a means for small to moderate size farm groups to remain in operation in the face of a concentrating agriculture production market in Illinois.

SMALL-TRACT HARVESTING

The feasibility of harvest operations on small tracts is often of concern when considering increased harvesting levels from NIPF timberland. Concerns about operability of small-tract harvesting are often centered on the perception that small NIPF tracts suffer from poor stocking levels, poor quality, and low growing-stock volume—qualities not sought after by would-be timber buyers and loggers. Conse-quently, a common notion is that successful harvesting operations on small NIPF tracts are limited to those tracts with only the highest quality species, and those with the largest growing-stock volumes.

In a study addressing the economic availability of timber for production in Tennessee, May and LeDoux (1992) elicited the characteristics of upland hardwood forests that were found to be economically feasible for harvest. The authors used the profitability model ECOST version 2 (Ledoux 1988) and FIA datasets to determine the average characteristics of 35-acre timber tracts located in Tennessee that were profitable to harvest. The model simulated clearcut harvesting systems using chainsaw felling and rubber-tire skidders. Though the willingness for NIPF owners in the Central States to use clearcut operations is in doubt (Roberts and others 1995), the study results provide a clear baseline for comparison of forest characteristics.

In their study, May and LeDoux (1992) found that the average 35-acre tract, profitable to harvest under moderate logging costs, has a mean tree diameter of 10.7 inches and a harvested growing-stock volume of 1,795.8 cubic feet/acre. While these average characteristics are most appropriate for Tennessee, the findings suggest that biological characteristics of economically feasible timber tracts are certainly attainable and not limited only to most productive NIPF tracts.

Studies of the characteristics of farmer-owned NIPF timberland have been completed in the states of Florida, South Carolina, and Virginia

(Thompson 1997, 1999; Thompson and Johnson 1996). The studies suggest that farmer landowners in each of the studies were most likely to own forest tracts between 11 to 50 acres in size—a range that includes the 35-acre tract size of May and LeDoux (1992). Volume of growing stock per acre was compared across six tract size classes in each of the southeastern studies (Thompson 1997, 1999; Thompson and Johnson 1996).

While significant difference in growing-stock volume was found when testing for differences among all tract size classes in the Thompson studies, no specific differences were established due the absence of post-hoc analysis in the report. However, little practical difference in the volume of growing stock per acre is evident between the 11 to 50, 51 to 100, and 101 to 200-acre tract size classes. If this is the case, little variation may exist in degree of stocking between less than 200-acre tract categories. This is particularly important when comparing the potential basal area/acre available from tracts of different acreage. While the total available harvest volume will certainly be greater in a cumulative sense for large forest tracts, the similarity in growing stock per acre may suggest that small tracts may be a competitive option for timber buyers in a market with a large number of less than 200-acre tracts.

The willingness of loggers and timber buyers to purchase timber from small NIPF tracts is certainly of paramount interest in determining any opportunities for Illinois farmers to capitalize on an expanding hardwood market. Kittredge and others (1996) surveyed loggers in Massachusetts to determine their attitudes toward tract characteristics and their associated willingness to bid on timber. Loggers operating in Massachusetts were found to be willing to bid, on average, on tracts of 5.3 acres with a total volume of only 17.1 mbf.

The biological factor that most affected their willingness to purchase a tract of timber was wood quality/value—suggesting that loggers find moderate to high-quality, small-acreage tracts valuable opportunities for harvesting (Kittredge and others 1996). These specific numbers are not directly applicable to Illinois due to differences between the two States in forest resource base and industry. However, the acceptance by loggers of small tracts of timber as viable timber harvests (assuming quality is high enough to offset costs) suggests loggers are willing to purchase small tracts associated with farmer-owned NIPF timberland.

METHODOLOGY

Two databases were queried to obtain information related to NIPF land on farms in Illinois: the Census of Agriculture database (AgCensus) (NASS 1997) and the Forest Inventory and Analysis database (FIA) (Forest Inventory and Analysis Research 2000). Farms are defined by the federal government as "a place which produced and sold, or normally would have produced and sold, $1,000 or more of agricultural products" (NASS 1997). The FIA data is collected on a rotating basis by USFS experiment station personnel and the database is administered by the Southern Research Station of the USFS. The AgCensus database is compiled every 5 years by the NASS via a census of farm operators. The FIA dataset does have some limitations related to the sampling accuracy and applicability of data to fine scale analysis; however, its usefulness in timber production studies is documented (May and LeDoux 1992, LeDoux and others 1995, 1996; Worthington and others 1996). All spatial analysis was completed using ArcView 3.2.

General forest data relating to farmer-owned NIPF land was collected from the AgCensus database for all counties in the State. This data, at the county level of measurement, included the number of farms in operation, farm acreage, number of farms with woodlots, and farm woodlot acreage. From this dataset a spatial database was created that identified for each county in the state the number of farms identified as having woodland, total farmer-owned woodland acreage, and the percentage of farm acreage dedicated to woodland. From the resulting spatial database, target regions within Illinois with the greatest occurrence and extent of farmer-owned NIPF lands were identified. Further analysis using specific forest data from the FIA datasets were completed for only these regions.

The FIA electronic database was queried for the following data reported at a county level of measurement:

1) timberland acreage,
2) volume of growing stock of all species, and
3) acreage of timberland with site classification greater than or equal to 85 cubic feet/acre/year.

Data were collected only for lands classified as being "farmer/rancher" ownership. No FIA data were available for the counties of Edwards, Richland, Mason, and Scott within the target area.

Average tract size of farmer-owned timberland was calculated for NIPF land in the target areas based upon AgCensus figures (NASS 1997). The average farmer-owned forest tract size per county was calculated by dividing the number of woodlot acres per county by the number of farms in the county reported as having a woodlot. While this is a coarse measurement, the calculation is useful for general analysis at a county level of measurement. The tract size calculation operates under the assumption that the timberland acreage reported by farmers in the AgCensus is based upon continuous forest tracts.

RESULTS AND DISCUSSION

General Characteristics of Illinois' Farmer-Owned NIPF Land

Illinois has 4 million acres of timberland as reported by the USDA Forest Service's Forest Inventory and Analysis (FIA) database (FIA 2000). Timberland is defined by the USFS as "forest tracts greater than 1 acre in size capable of producing greater than 20 cubic feet/acre/year of growing stock" (FIA 2000). Of the State's total timberland acreage, NIPF landowners own 3.6 million acres (greater than 90 percent of the total forest acreage). Nearly 45 percent (1.6 million acres) of the NIPF land is owned by Illinois' farmers.

The frequency and extent of farm woodlots, as reported in the AgCensus, is spatially divergent across the State. Those regions of the State where farmer-owned forest land is most prevalent (and thereby potential harvesting opportunities the greatest) are clearly evident when depicted spatially. Counties located in southern and western Illinois clearly have a greater frequency of farm woodlots (fig. 1), greater farm acreage (fig. 2), and a greater percentage of farmland dedicated to woodlots (fig. 3). The lack of forest land frequency and acreage throughout Champaign County and the central Illinois region is consistent with that Region's greater rowcrop productivity and topography more suitable for traditional agriculture. Farmers located in western and southern regions of Illinois have a greater opportunity to capitalize on existing forest acreage and are the target regions for further analysis.

Farmer-Owned NIPF Characteristics

Within the 56 county target region, farmers own 1.24 million acres of timberland. Thereby farmers—NIPF owners—in the 56 county target area own roughly 75 percent of the State's total farmer-owned forest acreage. The mean farmer-owned acreage per county is 24,000 acres is

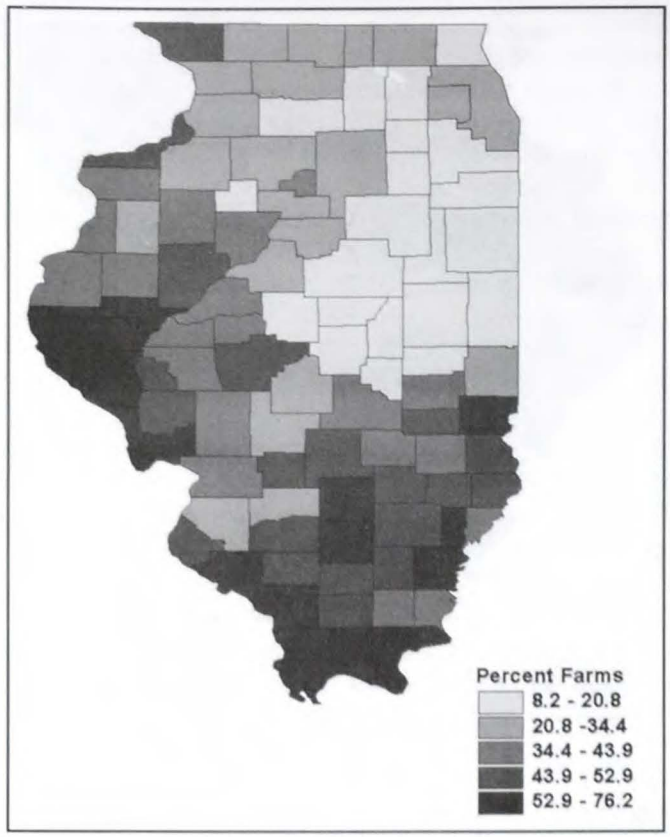

Figure 1.—*Percentage of operating farms having a "woodlot". (Source: NASS 1997)*

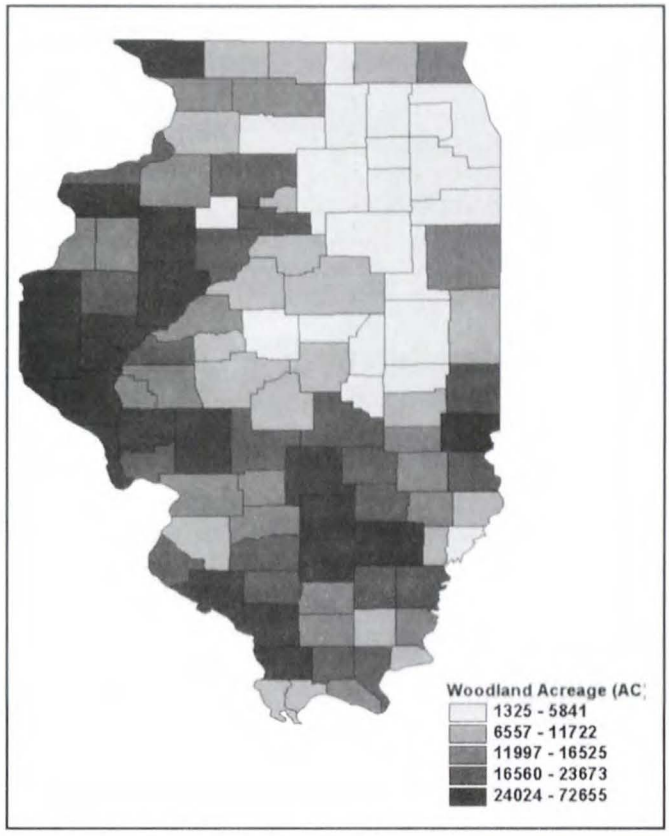

Figure 2.—*Acreage of farmer-owned "woodlots". (Source: NASS 1997)*

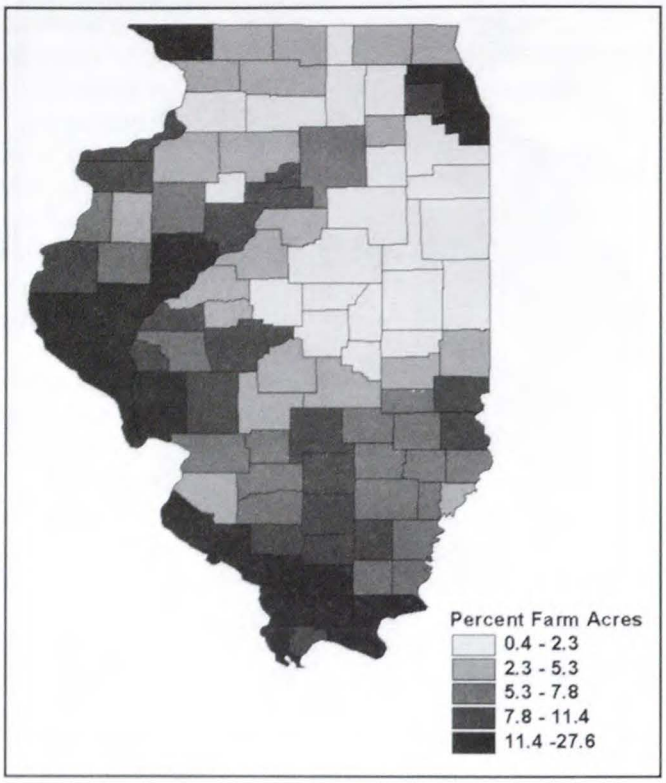

Figure 3.—*Percentage of farmland acreage classified as "woodlot". (Source: NASS 1997)*

highly variable with a standard deviation of 18,400 acres (fig. 4). Pike County had the greatest individual county acreage with 93,600 acres of timberland. St. Clair County had the lowest single-county acreage of 3,600 farmer-owned acres. Those counties with the highest total acreage are located along the I-57 corridor and the northwestern corner of the target region.

Total growing-stock volume within the region is 1,435 mmcf. Pike County with its large forest acreage has the greatest per county volume of growing stock with 111.7 mmcf (fig. 5). Likewise St. Clair County has the least growing-stock volume with only 4.5 mmcf. The mean growing stock per county is 27.6 mmcf (standard deviation of 20.7 mmcf).

The FIA database identifies site classification as the potential growth of fully-stocked natural stands (FIA 2000). That farmland acreage classified as being moderate to high production sites may have the greatest potential for timber production. Within the study area 535,900 farmer-owned acres are classified as being able to produce greater than 85 cubic feet/acre/year —43 percent of the timberland growing on farmer-owned acreage (fig. 6).

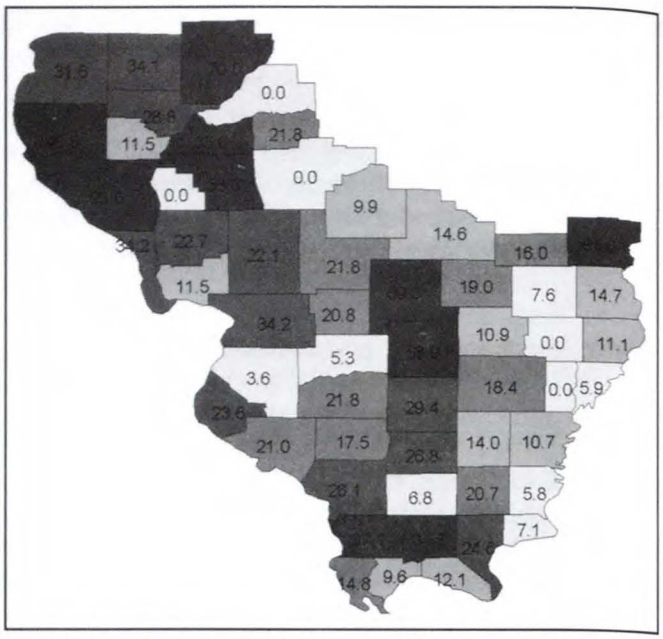

Figure 4.—*Farmer-owned timberland acreage (1000's) for counties in southern and western Illinois. (Source: FIA 2000)*

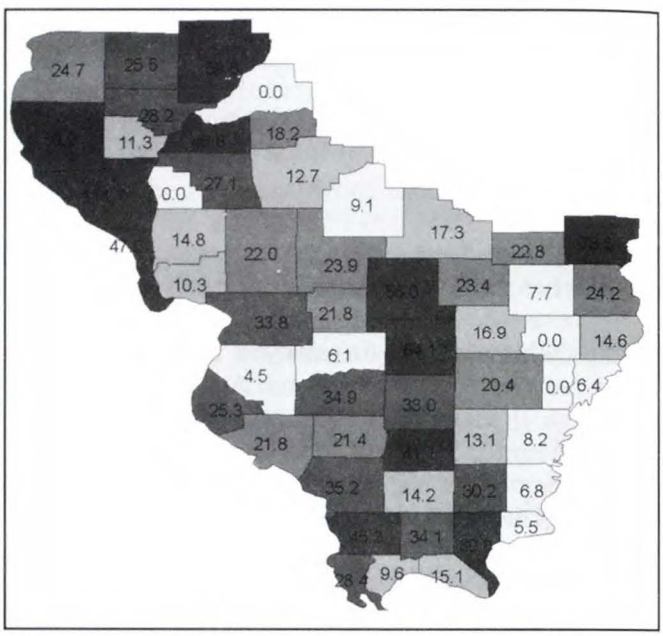

Figure 5.—*Volume of growing stock (mmcf) on farmer-owned timberland in southern and western Illinois. (Source: FIA 2000)*

A comparison of biological attributes for counties in the target region identifies the range of biological potential, at the county level, for farmer-owned NIPF timber harvesting. To control for differences in measurement unit, the values of the above biological components were standardized for each county. These standardized values were then summed to identify those counties with the largest value combinations of the three biological components. Figure 7 suggests that

while southern and western Illinois have biological characteristics most able to support increased timber harvesting from farmer-owned NIPF compared to the rest of Illinois, large variation in biological characteristics still exists. Successful implementation of programs directed at increasing harvesting on existing farmer-owned NIPF might depend upon identifying and targeting counties that have the greatest biological potential.

County tract size calculations for all southern Illinois counties yielded an average forest tract size per county of 70 acres (standard deviation 26 acres) (fig. 8). Mean farm size in the area is 363 acres. The greatest average tract size for farmer-owned acreage is 161 acres in Gallatin County in the southeastern portion of the State. Madison County has the smallest average tract size of only 38 acres. The average tract size owned by farmers per county reported is consistent with the average county tract size of all NIPF ownership as reported by the Illinois Forestry Development Council (IFDC 1999).

SUMMARY AND FUTURE RESEARCH

Farmers in southern and western Illinois who own existing forest tracts may experience positive economic impacts from timber harvesting operations. Within the target region, farmer-owned NIPF acreage, growing-stock volume, and site quality is of sufficient quantity to suggest that farmers in southern and western Illinois

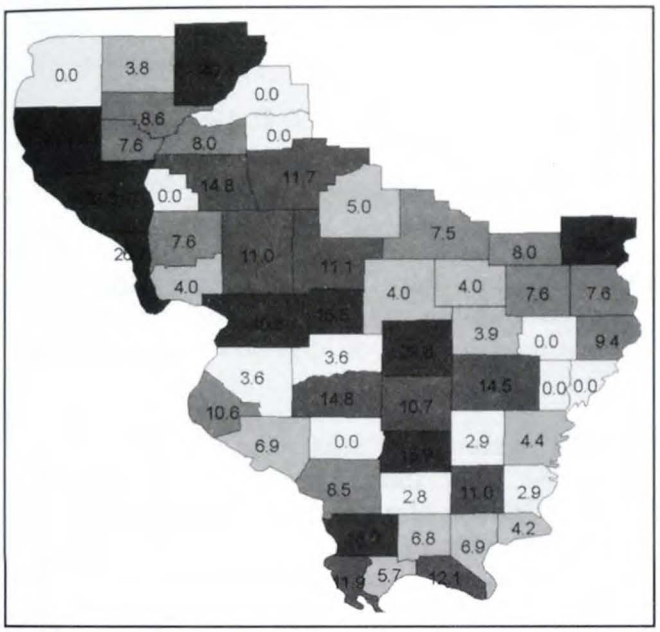

Figure 6.—*Acres (1000's) of farmer-owned timberland with a moderate to high site classification per county. (Source: FIA 2000)*

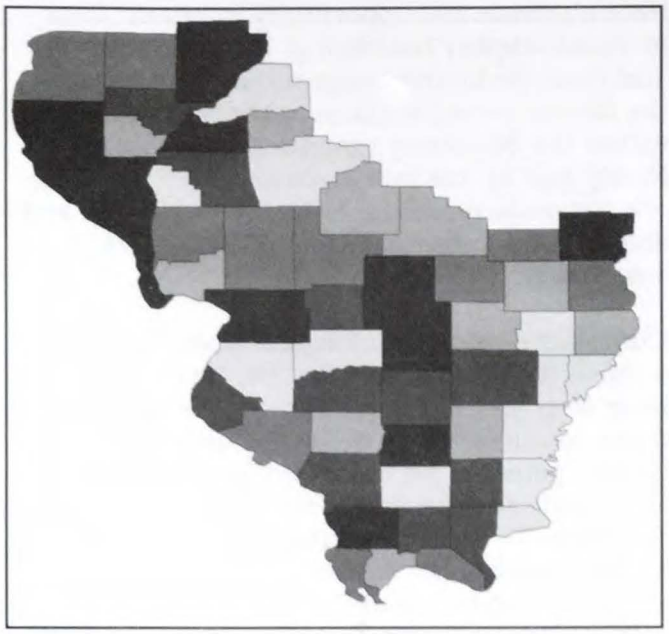

Figure 7.—*Counties with the greatest opportunity for timber harvesting on farmer-owned timberland based upon acreage, growing-stock volume, and moderate to high site classification values.*

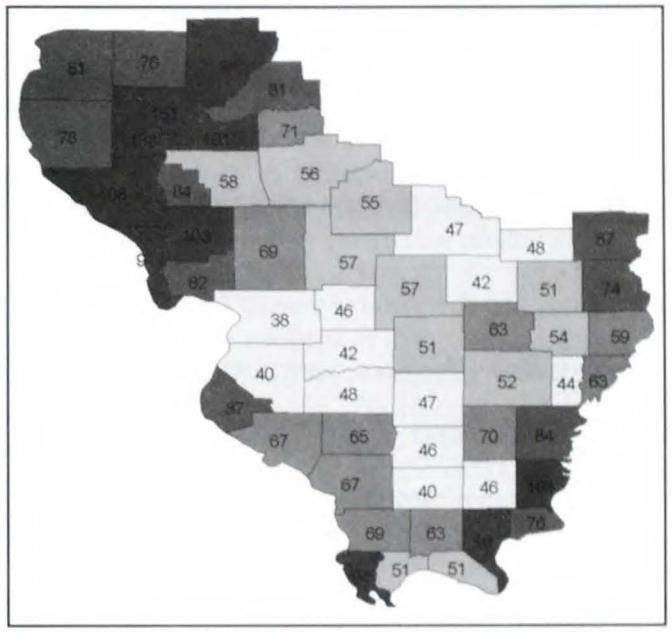

Figure 8.—*Average tract size of farmer-owned timberland in southern and western Illinois.*

have a substantial opportunity to benefit from increased timber harvesting. Forest Inventory and Analysis figures suggest that 75 percent of the farmer-owned acreage in the State occur within the 56 county target region identified. Nearly half of this farmer-owned NIPF in southern Illinois is moderate to highly productive and able to produce greater than 85 cubic feet/acre yearly.

Cumulative values of biological characteristics suggest that counties within the target area have divergent biological capabilities. However, those counties within the target area have greater potential for increased farmer-owned NIPF timber harvesting than other counties in the State. To insure that the productivity and sustainability of the resource is maintained, efforts should be taken that promote scientifically-based management of farmer-owned NIPF. In the absence of sound management the extensive farmer-owned NIPF resource may be lost to excessive timber harvesting or simple neglect.

Future activities will be directed toward
1) integrating this research with a model of timber mill procurement influence within Illinois to further identify areas within the State where farmers may have the greatest opportunity to sell timber from existing NIPF land;
2) analysis of the feasibility of timber harvesting on specific farmer-owned tracts or site types within southern and western Illinois; and
3) promoting the greater use of forest management plans by NIPF landowners, and particularly farmers, within the State.

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AN EXPERT SYSTEM FOR ESTIMATING PRODUCTION RATES AND COSTS FOR HARDWOOD GROUP-SELECTION HARVESTS

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ABSTRACT.—As forest managers shift their focus from stands to entire ecosystems alternative harvesting methods such as group selection are being used increasingly. Results of several field time and motion studies and simulation runs were incorporated into an expert system for estimating production rates and costs associated with harvests of group-selection units of various size by cable logging. This system also is useful for estimating production rates/costs for combinations of stand condition and unit size.

Public concern about clearcutting has resulted in the increased use of group-selection harvests. The small openings created by this technique allow landowners to harvest timber without impairing the visual quality of the site. Managers interested in containing logging costs also are concerned about the potential loss of production and profit in choosing group selection over conventional harvesting methods. The challenge for managers is to determine the harvesting cost associated with group-selection units of various size.

Group-selection entails harvesting small groups or clumps of trees in a somewhat random pattern across a stand. The goal is to capture financially mature trees as well as dead stems and those damaged by insects and disease, and to regenerate the stand. Major advantages of group selection over single-tree selection are that:

1) older mature trees can be harvested more economically and with less damage to the residual stand;
2) managers gain greater flexibility in increasing environmental conditions that favor successful reproduction; and
3) reproduction develops in well-defined, even-aged aggregations.

The latter is particularly important for the development of good tree form, especially among hardwood species (Smith 1981).

The openings or gaps created in the forest canopy by group selection increase the amount of desirable habitat for wildlife. Many wildlife species benefit from the combination of environmental conditions within and along the boundaries between the young reproduction and older adjacent trees. A wide array of protective cover is available in proximity to various food plants that may be created by various microclimatic conditions between the edges and the centers of the young groups.

The economic success of group-selection harvests of eastern hardwoods is dependent on product markets, tree species and quality, and logging costs (Bell 1989; Boucher and Hall 1989; Brummel 1992; Erickson and others 1992; LeDoux and others 1991, 1993; Hassler and others 2000). Several studies have attempted to define group-selection harvests and when they can be used (Roach 1974). Total harvesting costs generally increase as the size of the opening decreases. Results suggest that costs might be prohibitive when groups smaller than 0.50 acre are harvested by cable logging systems (LeDoux and others 1991).

Regeneration studies have shown that species composition can be affected by group selection (Walters and Nyland 1989, Dale and others 1995). Marquis (1965) found that the surrounding forest affects a greater amount of the total area of

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smaller than larger clearcuts. The increased competition for water and nutrients that results from these edge effects influences tree growth (Minckler and others 1973, Smith 1986, Dale and others 1995). Reductions in total height and merchantable growth of new regeneration were reported near the opening edge for 10 to 30 years following small group-selection cuts (Minckler and Woerheide 1965, Sander and Clark 1971, Willison 1981, Hilt 1985, Dale and others 1995).

LeDoux (1999) used regeneration data from Dale and others (1995) and harvesting data from LeDoux and others (1991) to determine the size of group-selection openings that would maximize financial yields for upland oak forests in the central hardwood region. Although the optimal size of opening should be 1.5 acres or larger to maximize financial yields (LeDoux 1999), managers find it necessary to use units of various size to meet a variety of objectives. In this paper, we describe an expert system that was developed for estimating production rates/costs associated with harvesting hardwood group-selection units of various size by cable logging.

METHODS

Time and motion and stand data from field studies (LeDoux and others 1991) were input into the THIN-PC (LeDoux and Butler 1981) simulation model to develop cost estimates for combinations of group-selection units of different size and stand and operating conditions. THIN-PC also can be used to simulate the harvests by cable logging systems. Specifically, the model evaluates how different diameter classes, stand densities, yarding efficiencies, external and lateral yarding distances, spatial log distributions, and prebunch-and-swing strategies affect production rates and related direct costs of harvesting group selection units of various size.

THIN-PC was run for more than 5,000 combinations of group-selection units of 0.25, 0.50, 1.00, 1.50, 2.00, 2.50, and 3.00 acre along with varying values for average diameter at breast height (DBH) of the cut stand, log length, average log size, average slope yarding distance, and log parameters resulting in alternative volumes removed per acre. The stand attributes, average log length and slope yarding distances, and volumes removed per acre for observed stands are shown in table 1. The results from these runs were stored in a database that formed the observed domain of the expert system.

Additional stand data from forest-model plots was used to develop a broader range for DBH and volume removed per acre (table 2). The average cut stand DBH ranged from 7.2 to 24 inches. Log lengths were 16 and 32 feet and tree length. Average log size ranged from 9.6 to 89.12 ft^3. Average slope yarding distance ranged

Table 1.—*Stand attributes, average log length, average slope yarding distance, and volume removed per acre for observed stands*

Average DBH	Average log length			Average slope yarding distance	Volume removed per acre
	16 feet	32 feet	Tree length		
Inches		Ft^3		Feet	Ft^3
9.57	8.0	12.6	22.4	50-950	817
11.02	10.6	16.5	32.5	50-950	1,327
11.23	11.0	17.3	31.3	50-950	1,223
11.90	12.4	19.4	38.0	50-950	1,493
12.16	12.9	20.2	37.0	50-950	1,614
12.44	13.5	20.8	39.1	50-950	2,711
12.48	13.6	21.3	41.8	50-950	1,493
12.67	14.0	21.6	40.5	50-950	2,699
12.93	14.6	22.9	43.0	50-950	2,008
13.20	15.2	23.5	43.9	50-950	2,728
13.87	16.8	30.0	48.9	50-950	2,852
13.92	16.9	26.6	50.0	50-950	2,519
14.48	18.3	28.2	54.5	50-950	2,776
14.50	18.4	29.4	57.1	50-950	1,556
17.82	27.7	44.9	89.8	50-950	3,052
17.91	28.0	44.6	89.2	50-950	3,390

Table 2.—*Stand data for forest model plots A2, B10, C8, D14, E13, F4, G4M, and H4M*

Forest-model plot	Average DBH	Merchantable volume
	Inches	Ft^3/acre
A2	7.2	2652
B10	8.1	2528
C8	9.1	3556
D14	11.6	3124
E13	12.7	4922
F4	16.8	6315
G4M	20.4	6466
H4M	24.0	6871

from 50 to 950 feet in increments of 100 feet. Simulated volumes cut per acre ranged from 600 to 6,300 ft^3. The results from these runs were stored in a database that formed the continuous domain of the expert system. These data and data from the observed stands were used to develop the interpolation equations that formed the knowledge base used in the second of four user-access modules.

EXPERT SYSTEM DESIGN

The expert system was designed to handle all possible combinations by dividing the system into four modules and linking them using VP-EXPERT (Sawyer 1989), a commercially available expert system development software shell. The shell uses the backward chaining method of inferencing and a production rule-based system of knowledge representation. The expert systems are fairly generic, extracting data from the databases as necessary.

Module 1

The primary purpose of Module 1 is to exploit the discrete observed database domain generated. It helps the user to query the system database and obtain production rates/costs for the observed stands. There are two database domains within the expert system, an existing observed database domain and a runtime database domain. The latter is the database developed during the execution of the third module. Module 3 is used to generate a database at runtime (LeDoux and Butler 1981). Executing THIN-PC for different combinations of input parameters creates this runtime database.

Results from THIN-PC are stored automatically in a new runtime database. The various data fields are filled using the output generated by executing THIN-PC. The production rate is used to calculate the cost using an equation provided by LeDoux and others (1991). The equation (for the Christy cable yarder) is of the form:

$$\text{Cost } (\$/ft^3) = ((((V_{oac} * AC)/PR) + 3.92) * MR)/(V_{oac} * AC),$$

where V_{oac} = the volume per acre in ft^3; AC = the unit size in acres; PR = simulated production rate in ft^3/hour; MR = machine rate/hour; and 3.92 = constant for setup, road and landing changes, and delays in hours.

Module 2

The primary objective of Module 2 is to use statistical relations to determine the production rates/costs for a variety of unit sizes and stand variables. For example, the database domain developed in Module 1 was for discrete levels of DBH (e.g., 9.57, 11.02, 11.23, etc.) and other stand variables. Module 2 uses statistical relationships to develop production rates/costs for continuous levels of input variables such as DBH of 9.57, 9.58, 9.59, etc.

Variables of interest included DBH, log length, average log size, unit size, slope yarding distance, and volume per acre. Suitable interpolation equations were developed for calculating the production rate based on the input parameters volume per acre, slope yarding distances, etc. Thirty different interpolation equations were developed to conduct the interpolations; these equations form the knowledge base for Module 2.

Module 3

The primary objective of Module 3 is to enable the user to execute THIN-PC within the expert-system environment and to create input files for executing THIN-PC. Another objective is to automate the process of database creation. The database domain created is called a discrete runtime database domain, which is exploited in Module 1. Another computer program retrieves the relevant data from the output file created by executing THIN-PC. The system then appends these values along with the output data obtained from executing THIN-PC onto the discrete runtime database. The system appends 10 records to the runtime database for each execution of the model. In this way, the runtime database is developed and various executable programs are executed within the expert-system environment.

Module 4

The primary objective of Module 4 is to allow the user to query the database for feasible operating conditions within a specified maximum affordable cost allowed. For example, users can

search the database for all combinations of operating conditions that will result in a yarding cost of $0.15 or less per ft3. Module 4 uses the graphical features and backward chaining mechanism of VP-Expert to locate all feasible scenarios. It allows the user to obtain multiple sets of conditions as the result of a query based on maximum affordable cost.

Figures 1 and 2 show the sets of conditions that would result in a maximum yarding cost of $0.15/ft3 or less. Figure 1 shows a stand with an average DBH of 12.44 inches and a volume removed of 2,711 ft3/acre. Group-selection units of 1.0 acre can be used only if the wood is yarded tree length and slope yarding distances are not more than 350 feet. Figure 2 shows a similar stand with an average DBH of 12.48 inches and volume removed of 1,493 ft3/acre. Group-selection units of 2.0 acres that are yarded with 32-foot logs will require short yarding distances (50 feet or less). If the same-size units are logged tree length, the slope yarding distance can be increased to 650 feet.

Generally, larger units results in more volume removed per entry, and larger logs are cheaper to harvest. Combinations that remove more volume/acre from larger units and that use tree-length yarding provide the largest set of feasible options. These combinations also allow the use of longer yarding distances. Another way to use this information is to fix the slope yarding distance at, say, 850 feet. The only feasible set in the 12.48 DBH stand would require units of 3 acres and that the wood be yarded tree length.

VALIDATION

The expert system was validated by comparing observed results (Module 1) with estimated results (Module 2) for numerous combinations. For comparison purposes, select combinations of observed values were input and Module 1 was used to develop estimates for discrete values. The same combinations of values were then entered into the system and Module 2 was used to develop estimates in a continuous mode. The results were comparable for all combinations tested. Table 3 is a sample listing of the validation tests. In every instance, the differences between observed and predicted values were within ±10 percent. On the basis of the results of these validation tests and the differences between observed and predicted values, we believe that the expert system can be used to develop cost and production information for group-selection harvests by cable-logging systems.

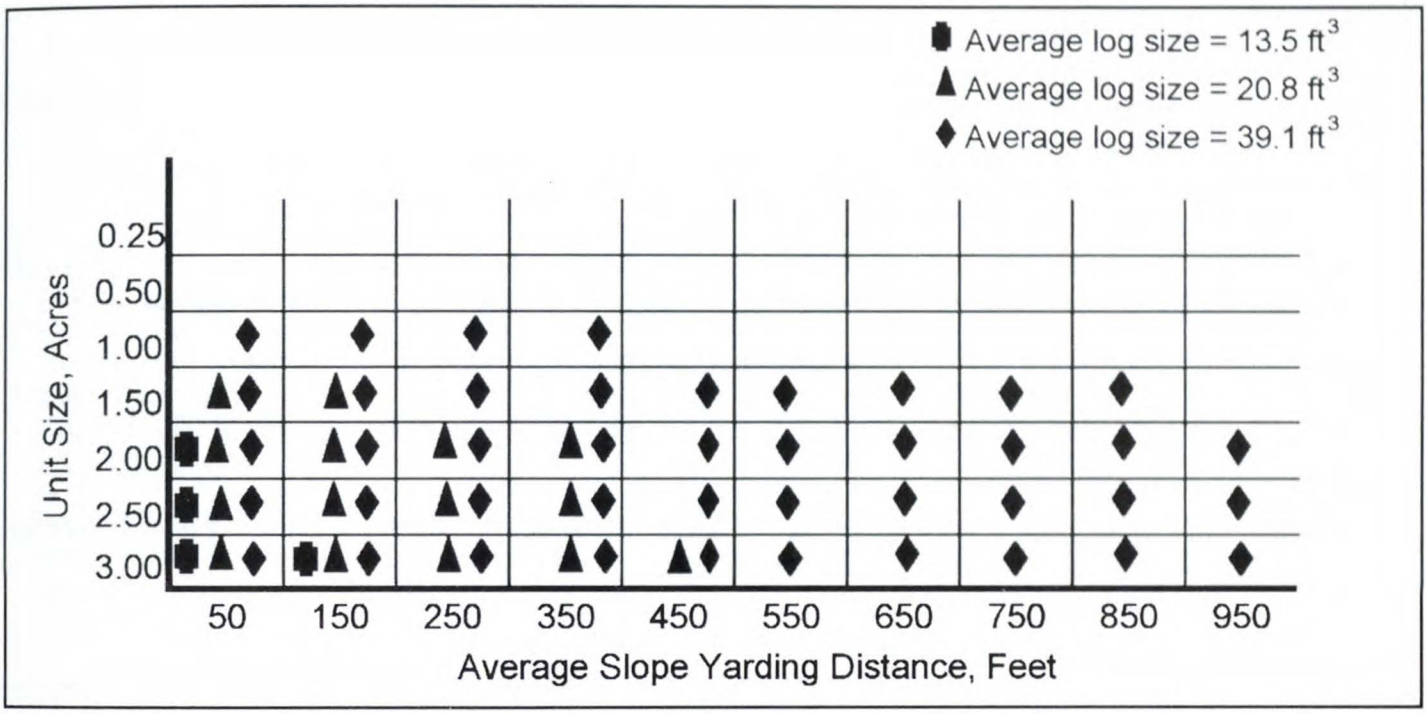

Figure 1.—*Set of combinations that would result in maximum yarding cost of $0.15/ft3 (average DBH cut = 12.44 inches; volume removed = 2,711 ft3).*

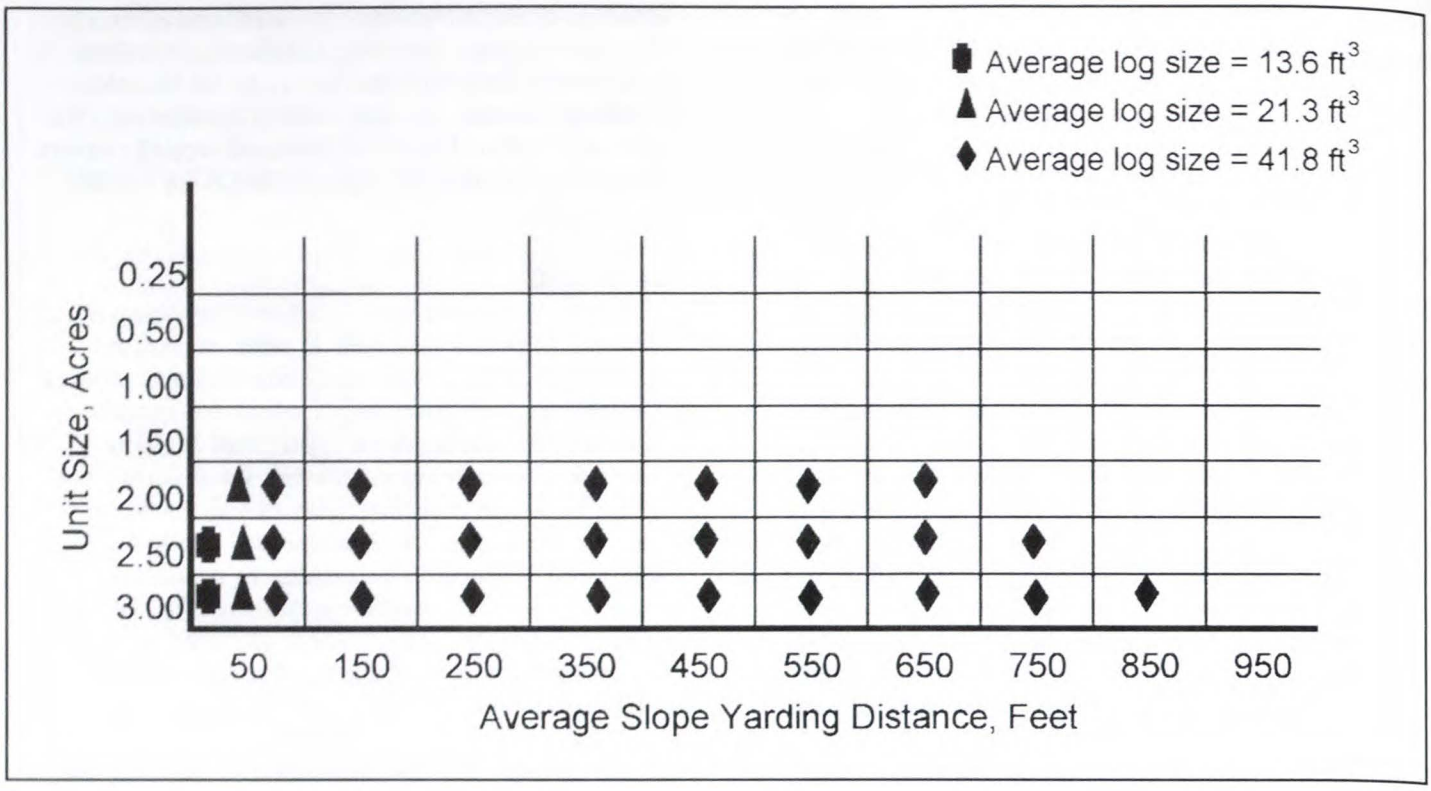

Figure 2.—*Set of combinations that would result in maximum yarding cost of $0.15/ft³ (average DBH cut = 12.48 inches; volume removed = 1,493 ft³).*

Table 3.—*Results of validation tests between observed and estimated values*

Test number	Unit size	Average log length	Average slope yarding distance	Average DBH cut	Volume/acre removed
	Acres	Feet	Feet	Inches	ft³/acre
1	0.25	32	350	9.57	817
2	0.25	32	350	11.02	1,327
3	0.50	32	650	11.02	1,327
4	1.50	16	750	14.48	2,776
5	3.00	16	950	17.91	3,390
6	3.00	Tree Length	550	17.91	3,390

Test number	Production rate and cost				% difference	
	Module 1		Module 2			
	Ft³ / hr	$ / Ft³	Ft³ / hr	$ / Ft³	Ft³ / hr	$ / Ft³
1	162.09	1.34	172.11	1.31	+6.18	-2.24
2	342.77	0.78	357.42	0.77	+4.27	-1.28
3	224.14	0.55	226.61	0.54	+1.10	-1.82
4	236.04	0.27	248.09	0.26	+5.11	-3.85
5	298.99	0.20	270.69	0.21	-9.47	+5.00
6	767.32	0.09	803.02	0.08	+4.65	-4.44

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ASSESSING TREE CARE PROFESSIONALS' AWARENESS AND KNOWLEDGE ABOUT THE ASIAN LONGHORNED BEETLE

Jason M. Hathaway, Cem M. Basman, and Susan C. Barro[1]

ABSTRACT.—The level of knowledge and awareness possessed by tree care professionals about the Asian Longhorned Beetle (ALB) is critical for the successful detection and eradication of this pest. A small sample of tree care professionals was surveyed about the ALB (from within the City of Chicago) in the summer of 2001. Results indicate that only 35 percent of survey respondents were "informed" (answered all questions correctly) with respect to the four ALB knowledge questions. Only 35 percent of respondents who reported having "high experience" dealing with pest infestations were "informed" with respect to the four ALB knowledge questions. An increased effort in educating professionals in the green industry may result in swift detection and eradication of the ALB and future pest infestations.

Each year a wide variety of organisms are unintentionally introduced into the United States primarily as a result of the increase in world trade (Haack and others 1997). According to the United States Congress (1993), some 4,500 exotic organisms are established in the United States. Of these organisms, more than 400 are insects that feed on trees and shrubs (Haack and others 1997). Many tree feeding exotics cause little apparent damage each year, while some have caused significant damage to forest ecosystems.

A recent and dangerous exotic insect to arrive in the United States is the Asian Longhorned Beetle (*Anoplophora glabripennis*). The Asian Longhorned Beetle (ALB) was first detected in the United States in New York City in August 1996 (Haack and others 1997). The eradication strategy for the ALB calls for removal of any tree found to be infested. According to a recent report (USDA 2001), 5,324 trees from within the New York City area had been removed because of ALB infestation. The ALB has also caused significant damage to urban trees within the greater Chicago area since its discovery there in the summer of 1998. The City of Chicago, along with state and federal partners, moved quickly to respond to the infestation. Chicago quarantined the movement of wood materials out of the most heavily infested areas. By the summer of 2001, the quarantined areas included the Chicago neighborhoods of Ravenswood, Addison, Summit, Park Ridge, and O'Hare Airport (fig. 1). As of September 2001, the infestations by the ALB had led to the removal of 1,523 trees from within the quarantined areas of Chicago (USDA 2001).

Without a quick response to the ALB infestation by the City of Chicago in cooperation with State and Federal agencies, the situation would have been much worse. According to estimations made by Nowak and others (2001), up to 2.5 million or 61 percent of all trees within the area would have been at risk, at an estimated cost of $1.2 billion to the City of Chicago if the infestation had gone unchecked. In a worst-case scenario where no quarantine restrictions exist and a projected spread rate of 3 km per year, the City of Chicago could become totally infested in only 5 years (Nowak and others 2001). The magnitude of this devastating threat has caused the eradication of the ALB to become a very high priority for the City of Chicago.

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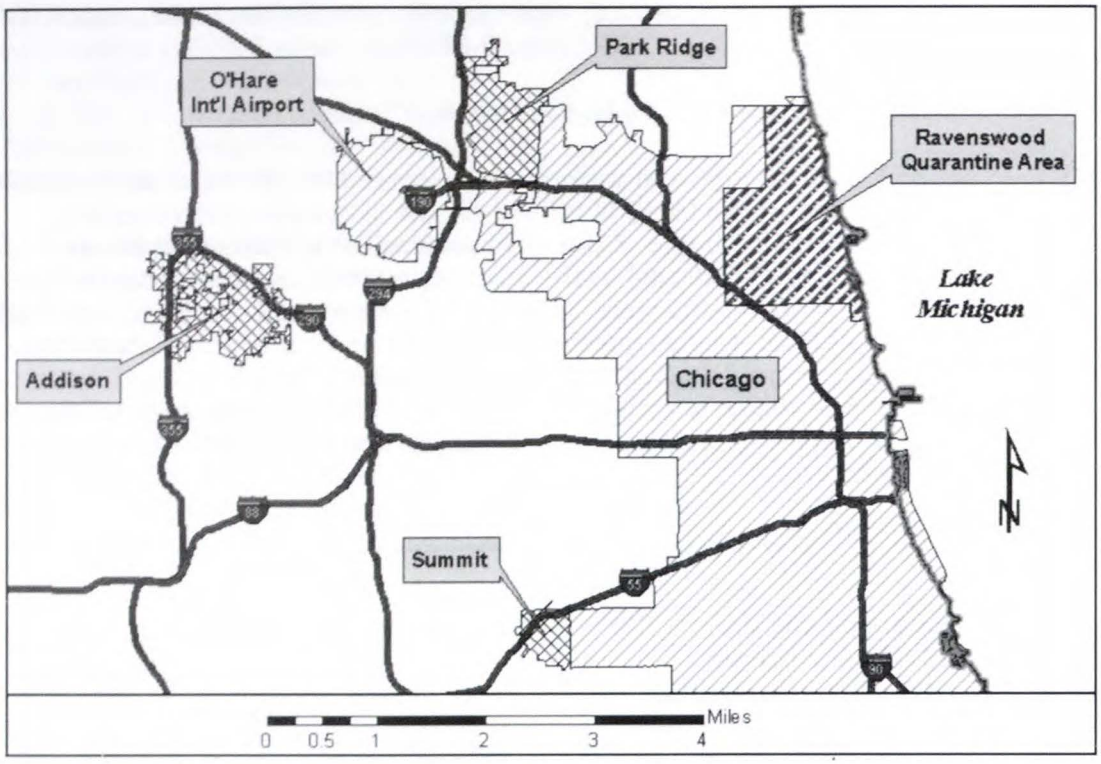

Figure 1.—*Quarantined areas for the Asian Longhorned Beetle in the Chicago-land area through the summer of 2001 (USDA 2001).*

Preventing the spread of ALB infestation to other major cities remains a concern for governmental agencies dealing with pest infestations. The ALB needs to be eradicated quickly from wherever it is detected. Tree care professionals, having day-to-day contact with a city's trees, can play an important role in the first line of defense against the beetle. Their effectiveness in this role is dependent on their knowledge and understanding of the biology and life history of the ALB, the signs of ALB infestation, and how to handle infested trees and woody material.

BIOLOGY AND INFESTATION

There are many host tree species to which the ALB is attracted. For example, in Chicago the ALB has been found feeding on the tree species: maple (*Acer*), horse chestnut (*Aesculus*), apple (*Malus*), chinaberry (*Melia*), mulberry (*Morus*), poplar (*Populus*), cherry (*Prunus*), pear (*Pyrus*), locust (*Robinia*), willow (*Salix*), elm (*Ulmus*), birch (*Betula*), rose of sharon (*Hibiscus*), ash (*Fraxinus*), and citrus (USDA 2001). As the beetle egg hatches in the bark, the larvae moves into the tree wood where it overwinters and feeds, and then emerges the following spring, signs of the ALB are left behind.

Detecting the evidence of an ALB infestation is critical for eventual eradication of the pest. Inspectors must examine individual trees for unique exit and entry holes of the insect. This can be cumbersome because the beetle typically chooses the top of the tree canopy as it begins its assault on the tree. Inspectors also look for piles of frass (sawdust) at the base of infested trees and in branch crotches, as well as sap leaking from wounds in the trees. Another sign of ALB activity is unseasonable yellowing or drooping of leaves when the weather has not been especially dry (Benson 1998).

While examining potential host trees for signs of ALB activity, inspectors may come across adult beetles working on limbs or trunks of infested trees. Being able to confirm sightings of adult beetles is another crucial method of detection. Adult ALB are 1.25 inches long with antennae that are as long as their bodies (Benson 1998). Each antenna is checkered black and white, and the body is shiny black with white spots. Larvae are cream-colored with distinct mandibles and mature to the same length as adult beetles (Ellis 1998).

After a tree has been found to be infested with an ALB, infested plant material must be removed and disposed of properly. Because of the lengthy egg-laying period, deep feeding habits within the tree by the larvae, and lack of any significant biological control agents, complete tree removal has been chosen as the best

strategy for eradication (Haack and others 1997). Trees that are removed are cut, chipped, and incinerated. This method of tree removal has proven to be very effective within the greater Chicago-land area.

EFFECTIVE COMMUNICATION NEEDS

The level of knowledge and awareness of ALB by tree care professionals may affect their ability to swiftly and successfully detect this pest. Informational and educational materials available to tree care professionals can play an important role in raising awareness and in turn detecting the beetle. Governmental agencies and organizations charged with eradicating the ALB need to provide useful and understandable information and educational materials about ALB to tree care professionals. Tree care professionals deal directly with the resource. They may be the first to be consulted by homeowners who suspect something is wrong with their tree. Ultimately a more informed cadre of tree care professionals may lead to more efficient detection and eventual eradication of the ALB.

Attempts to provide an effective education and information effort to a specific group about a natural resource issue can be challenging according to research by Bright (1994), as the success levels of the communication efforts are highly variable. The effectiveness of a communication effort can improve with carefully focusing communication strategies to the specific needs of a particular target group (Basman 1998).

Targeting educational efforts may begin by determining what a specific group currently knows about an issue (Slater 1992). For example, by understanding the extent and scope of knowledge held by tree care professionals about the ALB within the greater Chicago-land area, more effective educational materials could be developed by government agencies and organizations charged with disseminating information. There have been many endeavors by government agencies and organizations to disseminate information to the green industry, however, to date, no assessment of the tree care professionals' knowledge or beliefs about the ALB has been conducted.

The purpose of the current study was to assess tree care professionals' awareness and knowledge levels about the Asian Longhorned Beetle (ALB) within the greater Chicago-land area, and to determine the sources from which tree care professionals received information regarding the ALB, and the perceived credibility of these sources. Specifically, the study objectives were to:

1) Determine the various sources of ALB information utilized by tree care professionals and assess their perceived credibility and utility of the information sources.
2) Determine the various sources of ALB training made available to tree care professionals and assess their perceived credibility of these sources.
3) Assess tree care professionals' knowledge levels regarding four diagnostic signs of ALB infestations and develop a proficiency profile of their knowledge levels.
4) Assess the self-perceived experience levels of tree care professionals regarding pest infestations and compare to their proficiency levels about the ALB.
5) Develop suggestions for improving communication and education efforts by governmental agencies and organizations to tree care professionals regarding the detection and eradication of the ALB.

METHODS AND PROCEDURES

The study subjects were selected from tree care companies located in or adjacent to the ALB quarantined areas within the greater Chicago area by a non-random convenience method. A list of 118 tree care companies within or adjacent to these quarantined areas was compiled from telephone directory listings and an Internet search. In June 2001, each of the 118 tree care companies were contacted by phone. The researcher provided a brief explanation of the study and asked if the company would be willing to participate.

If the company agreed to take part in the study, they were mailed a survey. This occurred during the peak summer business time for tree care professionals when ALB issues would be most salient. A self-addressed pre-paid envelope was provided with the survey for the return of surveys upon completion.

Survey Instrument

The survey instrument for the current study evaluated respondents' knowledge about ALB and its life history, their beliefs and belief strengths about the status of the ALB infestation, the types of educational and training materials available from government sources, and the type of information and education delivery methods best suited for the tree care profession. Basic sociodemographic information was also collected.

Tree care professionals were asked four multiple choice questions to assess their knowledge regarding the ALB: "where sawdust created by

the ALB would most likely be located;" "the shape of ALB exit holes;" "the size of ALB exit holes;" and "what is the current most effective method of eradicating the ALB." To assess beliefs about the status of the ALB infestation in Chicago, respondents were asked four questions about their beliefs about the status of the ALB infestation in Chicago. A 7-point Likert-type scale was utilized to assess the belief scores of respondents that ranged from "strongly disagree or very detrimental" to "strongly agree or very beneficial," with "neither" as the midpoint of the scale.

To assess the certainty scores for the belief questions, a 7-point Likert-type scale was utilized that ranged from "very uncertain" to "very certain," with "neither" as the midpoint of the scale. According to Bright (1994) and Basman (1998), certainty about a belief is an indication of the strength by which these beliefs are held by individuals.

Self-perceived experience levels of respondents were assessed using a five-point scale that ranged from "no experience" to "much experience", with "average experience" measure as the midpoint of the scale. General sociodemographic information regarding individual and business backgrounds was also collected. Open-ended questions were also asked allowing respondents to make additional comments regarding the ALB infestation in the greater Chicago area. Respondents were provided a list of information sources and delivery systems (including those methods currently used by USDA-APHIS, USDA Forest Service, and City of Chicago) and asked to identify the "most useful" method in which they would like to receive information regarding the ALB.

Survey data were entered into SPSS Version 10 for analysis. Descriptive type data were collected as well as Chronbach's Alpha tests were run to determine reliability coefficients.

RESULTS

Of the 118 total companies contacted via telephone calls, 22 companies agreed to participate and 69 companies were left messages but did not respond, 13 did not wish to participate, and 14 were out of business at the time the survey was conducted. A total of 91 surveys (22 who agreed to participate and 69 left messages) were mailed to tree care companies. Thirty-four surveys were returned resulting in a 37 percent response rate.

Demographic Information

Tree care professionals located from within the Chicago-land area were used as respondents for the current study. Eighty-two percent of respondents were male and 15 percent of respondents were female. Ages of respondents ranged from 29 years old to 69 years old with the mean age being 48 years. Position titles listed by surveyed respondents were mainly owner/president at 53 percent of respondents. The remainder of job titles listed by survey respondents was: manager (18 percent), supervisor (9 percent), office manager (9 percent), sales (3 percent), climber (3 percent), and consulting arborist (3 percent). Seventy-seven percent of survey respondents reported they were fulltime employees.

Respondents reported they have worked in the tree care industry from 1 year to 44 years with 21 years as the mean number of years. More than half (55 percent) reported they were certified within the tree care industry. Of those who reported being certified, 27 percent were certified from the International Society of Arboriculture and 21 percent being certified from the Illinois Arborist Association. Additionally, the majority of survey respondents (70 percent) reported their highest level of education completed as either "some college" or as a "college graduate." The remainder of survey respondents reported their highest level of education completed as: "high school graduate" (10 percent), "trade or technical school" (10 percent) and "graduate or professional school" (10 percent).

ALB Information Sources

Survey respondents were asked to indicate whether or not they had received ALB information from seven different sources. They were also asked to report how credible they believed these sources to be. Eighty-eight percent of respondents indicated that they received information from "home television" and 65 percent of respondents received ALB information from the "USDA-APHIS" (table 1). The information received from the "USDA-APHIS" was reported as being the most credible (m = 4.57) The "Internet" was reported as the least utilized source of ALB information by tree care professionals, being used by less than 1 percent of respondents.

To assess source credibility, an index was created by summing respondents' scores for quality and value of the score. Reliability tests yielded Chronbach's Alpha scores of 0.83 to 0.99 for credibility index scores.

Table 1.—*Sources of information about the Asian Longhorned Beetle utilized by tree care professionals and their perceived credibility scores. (n=34)*

Information Source	Utilized by respondents Percentage	Alpha**	Credibility index n	Mean*	SD
Home television	88 %	30	3.79	0.94	0.83
Home newspaper	70 %	24	3.76	0.98	0.91
USDA-APHIS	65 %	22	4.57	0.48	0.96
Work newsletters	50 %	17	4.41	0.67	0.88
Home radio	50 %	17	3.56	1.25	0.95
Chicago/Dept. of Forestry	32 %	11	4.15	0.41	0.99
Work training	27 %	9	4.44	0.39	0.97
USDA Forest Service	12 %	4	4.38	0.47	0.99
Internet	—	1	3.00	—	—

* Mean scores based on 5 point credibility scale: 1 = very poor, 2 = poor, 3 = neutral, 4 = good, and 5 = excellent.
** Chronbach's Alpha reliability coefficient for credibility index (mean of "quality" and "value" scores).

Respondents were asked to report the preferred methods for receiving ALB information in the future from a provided list of methods or by selecting "other" and writing in a method (table 2). Nearly 60 percent of respondents would prefer to receive their ALB information in the future from "bulletins." Only 14 percent of survey respondents reported that they would prefer to receive ALB information in the future from the "Internet."

Survey respondents were asked to indicate whether or not they had received ALB training from four different organizations (USDA-APHIS, City of Chicago, USDA Forest Service, and place of employment) and to report on the perceived quality of this training. Nearly 30 percent of the respondents received ALB training from their "place of employment;" while 17 percent indicated they had received training from "USDA-APHIS." Only 2 respondents indicated receiving ALB training from the "City of Chicago" (table 3). ALB training provided by the "USDA-APHIS" was considered of greatest value with a mean score of 4.57. An index was created by summing respondents' scores for quality and value of the source. Reliability tests yielded high Chronbach's Alpha scores of 0.96 or above for credibility index scores.

Knowledge and Experience

Survey respondents were asked four multiple choice knowledge questions regarding ALB diagnostic signs to determine knowledge levels. The majority of tree care professionals answered the ALB diagnostic knowledge questions correctly (table 4). Sixty-one percent of respondents answered correctly to "sawdust location," 94

Table 2.—*Use of ALB and other pest related information. (n=34)*

Method	Frequency	Percentage
Bulletins	20	58 %
Brochure	12	35 %
Video tape	10	29 %
Reports	6	17 %
Internet/ Websites	5	14 %
Training sessions/ workshops	4	11 %
Seminars	3	<10 %
Email	3	<10 %
Company newsletter	2	<10 %
Magnets (refrigerator)	1	<10 %
Other[1]	5	14 %

[1]Included: mailed items; seminars from Illinois Arborist Association.

percent of respondents answered correctly to the "shape of ALB exit holes," 73 percent of respondents answered correctly to the "size of ALB exit holes," and 87 percent of respondents answered correctly to the "most effective eradication methods of the ALB."

A proficiency profile of tree care professionals' knowledge levels was created based on the total number of correct answers each respondent had to the knowledge questions. Survey respondents answers to the ALB knowledge questions were grouped into three new categories. Survey respondents who answered all four questions correctly were considered "informed." Survey respondents who answered two or three questions correctly were labeled "inconclusive" with respect to their proficiency. Although getting

Table 3.—*Asian Longhorned Beetle training sources utilized by tree care professionals and their perceived quality scores. (n=34)*

Training source	Utilized by respondents		Perceived quality index		
	Percentage	Alpha**	n	Mean*	SD
Work training	27 %	9	4.44	0.39	0.97
USDA-APHIS	17 %	6	4.57	0.48	0.96
Chicago Forestry	—	2	4.44	—	—
USDA Forest Service	—	1	5.00	—	—

* Mean scores based on 5 point quality scale: 1 = poor, 2 = fair, 3 = average, 4 = good, and 5 = excellent.
** Chronbach's Alpha reliability coefficient for perceived quality index (mean of "quality" and "value" scores).

Table 4.—*Assessment of tree care professionals' answers to knowledge questions about the Asian Longhorned Beetle. (n=34)*

Knowledge question	Answer correctly	
	Frequency	Percent
Sawdust (frass) location	20	61 %
ALB exit hole shapes	31	94 %
ALB exit hole size	24	73 %
Effective eradication method	28	87 %

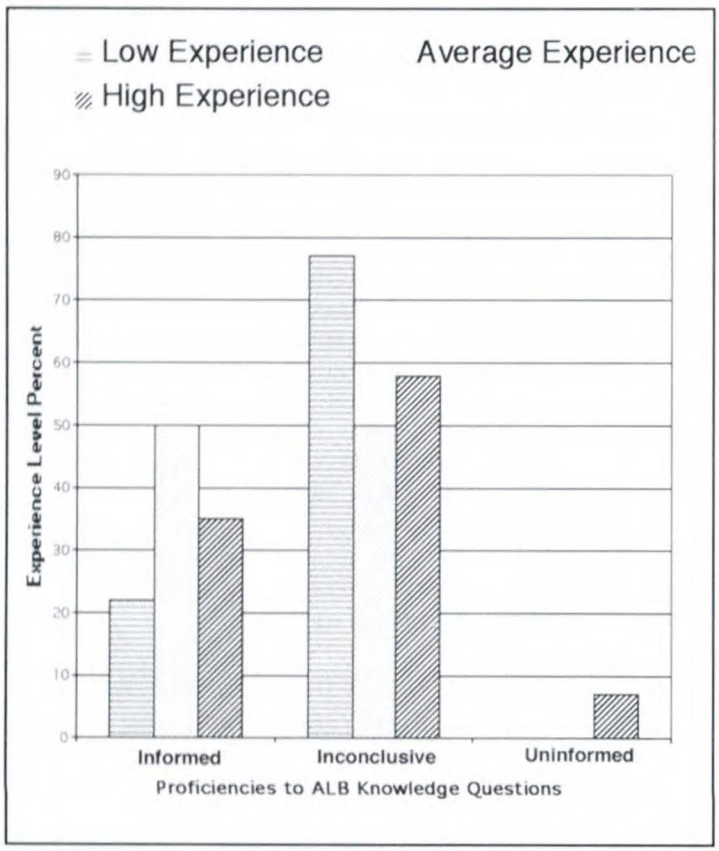

Figure 2.—*Percent of self-perceived experience levels regarding pest infestations by tree care professionals knowledge proficiency levels about the Asian Longhorned Beetle.*

three or four questions correct may seem to indicate substantial knowledge, any sign of ALB not detected could lead to breaking quarantines. Those who answered zero questions correctly were considered "uninformed." Respondents for this study were 35 percent (n = 12) "informed", 59 percent (n = 20) "inconclusive", and only one respondent was "uninformed" about the ALB (fig 2).

Survey respondents were asked to report their self-perceived experience levels regarding pest infestations with response choices of: no experience, little experience, average experience, a fair amount of experience, and much experience. Respondents' proficiency levels (as assessed through response to ALB knowledge questions) were then compared to experience levels. The larger majority (77 percent) of respondents who reported having a low level of experience regarding pest infestations fell into the "inconclusive" knowledge category (fig. 2). Respondents indicating an average amount of experience regarding pest infestations were either "informed" (50 percent) or "inconclusive" (50 percent). Respondents who reported having a high level of experience regarding pest infestations had a proficiency level of "inconclusive" to ALB knowledge questions making up 58 percent of survey respondents within that group.

CONCLUSIONS

Our study was a preliminary study looking at tree care professionals' knowledge and awareness levels regarding the ALB or other pest infestations. Although our study had a relatively low response rate, it can still serve as an initial step in learning more about this important group of people in the battle against the ALB. The study focus on awareness, knowledge, and information sources was targeted at identifying places where the communication process breaks down. With this knowledge we can take steps to overcome these communication barriers. The

current study indicates the largest amount of information tree care professionals receive about the ALB comes from local media sources (television and newspaper).

While the volume of information received by tree care professionals from the USDA-APHIS was not as great as that received from media sources, it was considered the most credible source. The tree care professionals we heard from are not currently utilizing the Internet to obtain ALB or other pest related information. This may indicate that tree care professionals do not have access or are unaware of information available from the Internet. Rather, respondents to this survey reported that they would prefer to receive ALB and other pest related information in the future mainly from brochures and bulletins. This has important implications for agencies such as APHIS, since more and more organizations are moving information to web based access and reducing or eliminating more traditional information venues.

Survey respondents reported their training was most frequently obtained from their place of employment. The number of tree care companies who received training from the City of Chicago (which is only available upon request) was very minimal. This may indicate that professionals from within the green industry are not aware of the training services offered by the city or do not have the interest or time to receive such training.

Slightly more than a third of respondents answered all four knowledge questions correctly. An increased effort from agencies in charge of dealing with the ALB and other pest infestations in training professionals in the green industry may increase the overall knowledge and awareness of tree care professionals insuring that the ALB and other pest outbreaks are swiftly contained and properly exposed of in the future.

Tree care professionals are challenging groups to communicate with because of the changeover of tree care companies (as shown by 13 percent out of business) and the diversity of tree care companies (small companies versus large companies) evidenced by the survey respondents. Tree care professionals are in a key position to make a difference in the war on the ALB and state and federal agencies charged with handling the ALB need to know how to best communicate with the green industry to 'win the war'.

ACKNOWLEDGMENTS

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FOREST PRODUCTIVITY: AN INTEGRATED RESEARCH AND DEVELOPMENT PROGRAM

Daniel C. Dey, Thomas R. Crow, and Don E. Riemenschneider[1]

ABSTRACT.—In 2000, the North Central Research Station initiated the Forest Productivity Integrated Research Program (North Central Research Station 2001). This program combines the efforts of scientists from across the Station's 13 research work units to examine the current condition of the forests in the North Central Region and their prospects for producing wood and fiber. The overall goal is to develop the scientific knowledge and tools needed to increase and maintain forest productivity and the supply of forest products in the North Central Region. The challenge is to optimize productivity and availability of forest products using sustainable silvicultural systems that are socially acceptable, economically viable, and ecologically feasible.

The main element of the poster was a listing of several research projects related to the forest productivity program. The central graphic showed interconnections among 31 studies and lead researchers within the NC station with the projects grouped into seven broad categories (Assessment, Riparian, Management and Silviculture, Intensive Culture, Management Effects, Environment, and Fire). However, many of these projects could be assigned to more than one category. For example, a study of the impacts of management on seasonal ponds could be considered appropriately placed under the silviculture, riparian, or management effects categories.

Recently, as the result of a with-in station call for proposals to further the goal of integrated research, 10 new efforts were funded from among 41 proposals submitted. Five of the proposals, including studies with ecological and social emphasis, have direct implications for forest productivity in the North Central Region.

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EFFECT OF ACORN MASS AND SIZE, AND EARLY SHOOT GROWTH ON ONE-YEAR OLD CONTAINER-GROWN RPM™ OAK SEEDLINGS

Benjamin C. Grossman, Michael A. Gold, and Daniel C. Dey[1]

ABSTRACT.—The purpose of this study was to evaluate the influence of seed source, acorn mass and size, and early shoot growth on morphological traits of 1-year-old oak seedlings. Forested bottomlands in the Midwest have been greatly reduced in the past 150 years as a result of conversion to agriculture and recent catastrophic floods. Due to a lack of available hard mast, intense competition from other riparian pioneer species and heavy animal pressure, natural regeneration and bare-rooted plantings of oak and other hard mast species along the lower Missouri River have had limited success. One viable option may lie in the use of root production method (RPM™) container-grown oak seedlings. Analysis of main treatment effects (i.e., heavy vs. light acorns) determined that *Quercus* x *schuettei* seedlings originating from heavy acorns were significantly larger than seedlings from light acorns. Main treatment effects on *Quercus bicolor* were analyzed and small acorns were found to have significantly larger seedling height and root volume than large acorns. Based on our findings, when seedlings are grown under optimal conditions for the production of high quality nursery seedlings, it is advantageous to retain and sow all acorns without screening for size or weight.

Presettlement floodplain forests along the Missouri and Mississippi Rivers were dominated by numerous tree species, including several hard mast species such as pecan (*Carya illinoensis* K. Koch), swamp white oak (*Quercus bicolor* Willd.), pin oak (*Q. palustris* Muenchh.), shellbark hickory (*C. laciniosa* Loud.), and black walnut (*Juglans nigra* L.) (Kozlowski 1984, Yin and Nelson 1995, Scott and others 1996). Since the 1800s, 70 to 90 percent of floodplain forest habitat has been lost in the continental United States due to natural changes in the river's course, land clearing for agriculture, and the construction of dams and levees to protect valuable cropland and aid navigation (Knutson and Klaas 1995). These modifications have caused the reduction and localized disappearance of hard mast bottomland tree species (Yin and Nelson 1995).

Floodplain forests stabilize river banks, protect levees, and reduce flood impacts (Low 1994, Dwyer and others 1997, Geyer and others 2000). Floodplain forests also sequester carbon and reduce nutrient run-off by absorbing excess fertilizer applied to adjoining cropland (Sparks 1995). Furthermore, floodplain forests provide a diversity of habitat for fish and wildlife (Sparks 1995, Dwyer and others 1997, Yin and others 1997). Late-succession tree species, such as oak, provide mast for large and small mammals (Van Dersal 1938) as well as numerous species of birds (Hirsch and Segelquist 1978, Twedt and others 1999).

Although the Great Flood of 1993 degraded over 800,000 acres of cropland in the lower Missouri River floodplain by deposition of sand and scouring of soils, there now exists excellent opportunities for reestablishing hard mast species in floodplain forests (Low 1994). Hard mast-producing tree species such as swamp white oak, bur oak, pin oak, and pecan are important for numerous species of aquatic and terrestrial wildlife including deer, turkey, and waterfowl. These same species have difficulty

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becoming established due to a lack of locally available hard mast seed, intense competition from pioneer riparian tree species [e.g., silver maple (*Acer saccharinum* L.) and cottonwood (*Platanus occidentalis* L.)] and aggressive regrowth of annual weed species, inherent slow juvenile growth of hard mast species, continued flooding and deer herbivory (Buckley and others 1998, Dalrymple personal communication). An alternative to traditional bottomland reforestation with direct seeding or bare-root seedlings is the use of large containerized nursery seedlings produced under the root production method (RPM™) (Lovelace 1998).

ROOT PRODUCTION METHOD™

In the Upper Mississippi and Lower Missouri River floodplains, RPM™ seedlings have considerable advantages over traditional bare-rooted nursery stock including improved growth and survival, ability to be fall planted, terminal tips that reach above seasonal flood levels and browse height (≥ 5 feet), large basal stem diameter (≥ 0.6 inch), precocious flowering and mast production in the third or fourth year. Dey and others (2001) found that 4 percent of 30-month-old swamp white oak grown as RPM™ produced sound acorns the first year after planting in former cropfields along the Missouri River.

Selection of acorns from a superior mother tree can aid the production of desirable planting stock (Buchschacher and others 1991; Kormanik and others 1990, 1998, 2003). Most research on the effect of acorn size within species has shown that as acorn mass increases, seedling root collar diameter, height, leaf area, biomass, survival, and rate of emergence increases (Bonfil 1998, Ke and Werger 1999), and could be due to a greater amount of food reserves in the seed and cotyledons of heavier acorns (Tripathi and Khan 1990).

A few studies, however, have found no relationship between acorn size and seedling growth and morphology (Long and Jones 1996). If larger acorns emerge earlier and develop into larger seedlings, they may have a greater probability of reaching reproductive size in a competitive environment than smaller acorns (Seiwa 2000), but under competition free environments such as in a nursery, acorn size may be of little advantage.

Nursery planting stock is typically graded to determine acceptable size and evaluate competitiveness once outplanted. These criteria include height and root collar diameter (RCD) (Belanger and McAlpine 1975, Johnson 1984), number of first-order lateral roots (FOLR) (Kormanik and others 1998, Schultz and Thompson 1990) and root:shoot ratio (Romero and others 1986). More recently, root biomass has been considered one of the most important factors in the survival and establishment of nursery seedlings (Swaminath and Ravindran 1990) and therefore more attention has been given to producing seedlings with large, intact root systems.

A trees' root system has the role of anchorage (Richards 1983, Schultz and Thompson 1992), storage of nutrients (Richards 1983; Bowen 1984, 1985; Schultz and Thompson 1992), water and salt absorption (May and others 1965, Aung 1974, Nielsen 1974, Richards 1983, Shultz and Thompson 1992), synthesis of nitrogenous compounds (Perry 1982), and production of hormones (Aung 1974, Richards 1983, Schultz and Thompson 1992). The root system can be modified by a number of cultural techniques. For example, basal root pruning in the nursery seedbed can increase the number of lateral roots and root tips (Schultz 1989; Schultz and Thompson 1990, 1992) that are important for seedling establishment (Struve 1990) and recommended for producing high quality planting stock. Air pruning of roots is another technique that is used in the RPM™ process to produce a large, dense, fibrous root system (Lovelace 1998).

Our overall purpose was to establish base line data on early morphological characteristics of bottomland oak species that may lead to precocious flowering in seedlings produced by the RPM™ process. Specific objectives of this research were

1) to determine the effect of acorn size and mass, and early seedling growth on morphological characteristics of 1-year-old seedlings and
2) to determine if a population of oak hybrids pre-selected for early and abundant acorn production would perform better than a randomly selected population of acorns when propagated under the RPM™ system.

MATERIALS AND METHODS

Acorns were collected in the fall of 1999 from two separate sources. One population consisted of randomly collected *Q. bicolor* acorns from Saline County, Illinois. The other acorns originated from Lincoln County, Missouri, from a population of half-sibling *Q. bicolor* x *macrocarpa* (Michx.), i.e., *Q.* x *schuettei* (Trel.). In the normal RPM™ process, acorns are graded by mass and diameter (Lovelace 1998). An aspirator is used to separate and discard the lightest

acorns below a pre-determined specific gravity, which are not considered sound. Remaining acorns are separated into two size classes, heavy and light, again based on pre-determined specific gravities.

Heavy and light acorns are then rolled across a plate with two different sized holes and separated into small and large size classes. Only the heaviest and largest acorns are stratified. In February, acorns are germinated in heated greenhouses in mesh-bottomed trays that allow for air pruning of the roots. After completion of the first shoot flush, seedlings are graded by precocious shoot growth and only the fastest growing seedlings (approximately the tallest 50 percent of germinates) continue in the RPM™ process.

After a series of transplants into increasingly larger bottomless containers placed on raised benches, seedlings are finally potted in shallow 3- or 5-gallon pots with a growth medium of rice hulls, pine bark, and sand that has 35 percent air space. Seedlings are then placed outside for the remainder of the growing season. Typical RPM™ seedlings attain heights of 4 feet or taller after 210 days of growth and they develop dense, fibrous root systems high on the root collar.

For this research, acorns were propagated into seedlings following the RPM™ process with the following important exceptions: as acorns and seedlings were graded, all seed and plant materials were separated and retained in their respective size classes instead of being discarded (i.e., both large and small seed, heavy and light seed, and precocious and non-precocious seedlings were retained). This resulted in eight treatments that represented all combinations of acorn mass (heavy or light), acorn size (large or small), and shoot growth (precocious or not).

Due to poor germination we had fewer seedlings than expected for some treatments and were forced to use two different experimental designs for each population. A 3 x 3 balanced lattice square design was used to examine all eight treatments (table 1). The *Q.* x *schuettei* population had four replications, each observation being the mean of two seedlings within a block. The *Q. bicolor* population also had four replications, but had fewer seedlings due to poor germination, and therefore, each observation was from one seedling. A 4 x 4 balanced Latin square design was used with only the four precocious treatments with one replication per population (table 2). Each block within each design contained three seedlings, from which a statistical mean was derived to obtain one observation.

Table 1.—*Partial ANOVA for balanced lattice square design*

SOURCE	DF
Replications	3
Treatments (adj.)	8
Rows (adj.)	8
Columns (adj.)	8
Error	8
Total	35

Table 2.—*Partial ANOVA for balanced Latin square design*

SOURCE	DF
Row	3
Columns	3
Treatments	3
Error	6
Total	15

At the end of the growing season, seedlings were destructively sampled and measurements were taken on seedling characteristics. Variables measured included root collar diameter (measured 2.5 cm above the root collar), height, root volume, root and shoot dry mass, and number of flushes. Root volume was measured by the displacement method described by Böhm (1979). Seedling mass was recorded after seedlings were oven dried and weighed on a top loading balance. SAS Version 8.2 (SAS Institute, Cary, NC) was used to conduct the analysis of variance, to test hypotheses relating acorn mass, acorn size, and initial shoot growth to first-year RPM™ seedling morphology, and to determine least significant differences (LSD) between treatment means.

RESULTS

Precocious shoot growth was not significantly related to first-year root or shoot size or morphology in either *Quercus* populations based on the Latin square design. Analysis of least significant differences showed that, seedlings from the light-small acorn treatment were consistently larger than any other acorn mass-size treatment combination (fig. 1).

Heavy *Quercus* x *schuettei* seed produced first-year seedlings with significantly larger root mass and volume, and shoot length and mass that did seedlings originating from lighter seed (table 3). Other researchers have found that seed mass is positively correlated or significantly related to early seedling growth (i.e., Eytingen 1917, Korstian 1927, McComb 1934, Rice and others 1993, Bonfil 1998).

A noted exception to the above occurred in the seed size treatment in the *Quercus bicolor* population, where seedlings from smaller acorns had

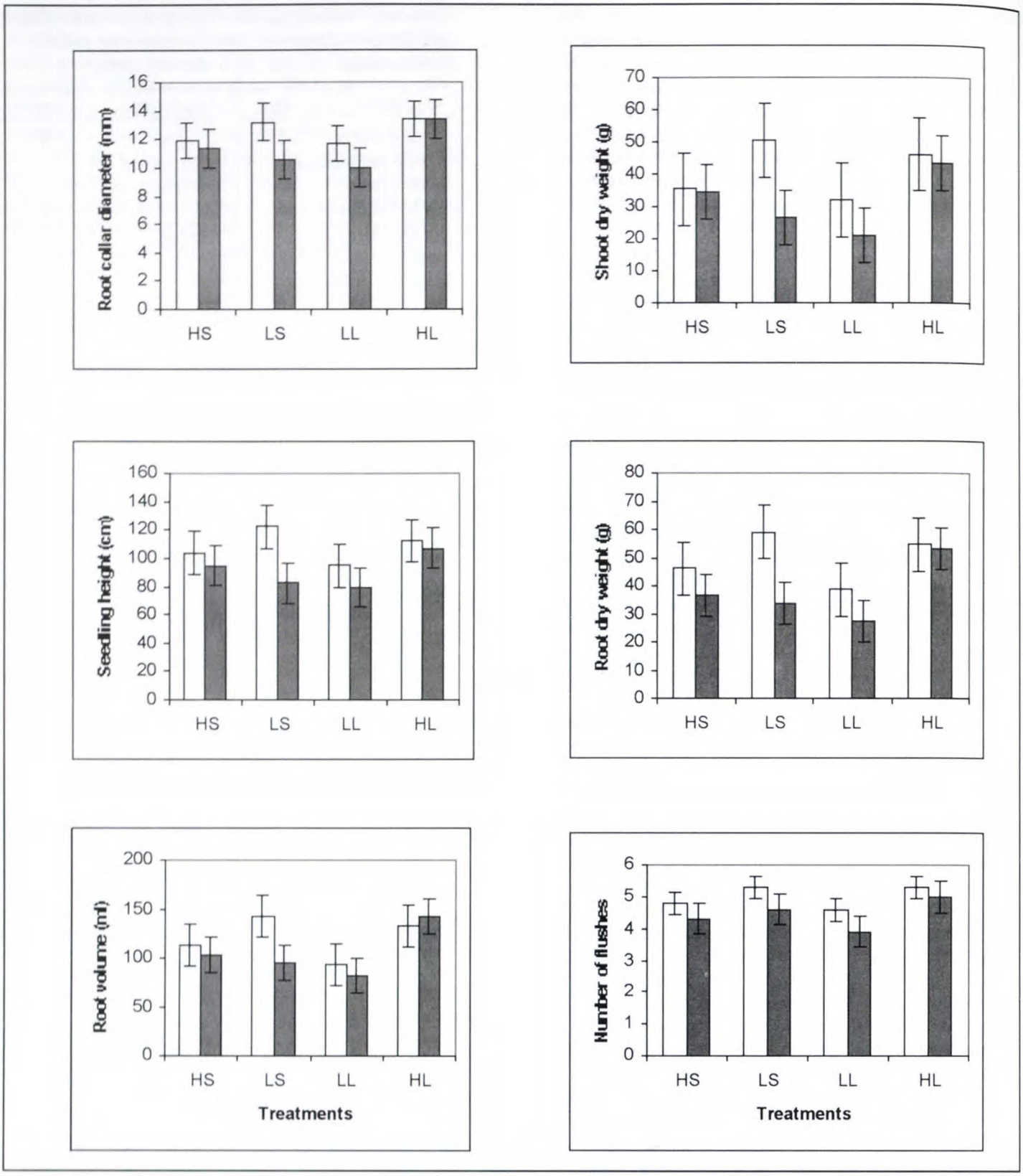

Figure 1.—*The effect of acorn mass and size and early growth on first-year morphology of RPM™ container-grown seedlings from two* Quercus *populations; Treatments: HS = heavy-small, HL = heavy-large, LL = light-large, LS = light-small. Means +/- standard error. Open bars =* Quercus bicolor*; shaded =* Q. *x* schuettei.

significantly greater root mass and volume, and shoot mass and size than seedlings from larger acorns (table 4). The small acorns from the *Q. bicolor* population were found to have significantly larger values for seedling height and root volume than large acorns. There was no other significantly different main or two-way interaction effects (i.e., heavy-large vs. heavy-small) found.

Significant differences were found among all eight treatments (i.e., all combinations of acorn mass and size, and precocious shoot growth) that were analyzed with the lattice square design from both *Quercus* populations, but these differences were not consistent for one particular treatment with one exception (fig. 2). For example, the heavy-small-precocious treatment from the *Q. bicolor* population was consistently, but not always, significantly larger than the other treatments from that population.

Q. bicolor seedlings from public collection were consistently, but not always, significantly larger than the hybrid *Q.* x *schuettei* seedlings (figs. 1 and 2). Overall, however, there were no significant differences in seedling morphology between *Q. bicolor* and *Q.* x *schuettei* seedlings after one growing season. *Q. bicolor* seedlings did have a greater amount of variation as indicated by the large standard errors than did the half-sibling *Q.* x *schuettei* seedlings.

DISCUSSION

In previous studies, Rodger (1920) found no effect of acorn mass and size on seedling characteristics, but his sample was too small to make definitive conclusions. Long and Jones (1996) also found no effect of acorn mass and size on seedling characteristics, but suggests that poor control over acorn moisture content (in their study and others) and genotypic variation on seed size could have influenced their results. Studies that involved the removal of a portion of the acorn prior to sowing, or removal of the cotyledons shortly after germination, found that seedling growth was more affected by poor soil nutrition than by loss of acorn mass (Sonesson 1994, Anderson and Frost 1996). Furthermore, it has been speculated that acorn size may be more important in attracting wildlife for seed dispersal than for first-year-growth (Stiles 1980, Anderson and Frost 1996).

Conflicting results may also be due to differences in propagation methods. Traditionally seedlings have been grown in seedbeds as bare-rooted nursery stock under less than ideal conditions for rapid growth. Containerized RPM™ seedlings grown in the greenhouse under optimal growing conditions are under no stress and are absent of competition relative to seedlings grown in the wild or in nursery beds. Larger seeds may have advantages over smaller seeds if both are grown with competition (Wulff 1986, Bonfil 1998), but under low stress and competition-free environments, little appears to be gained from larger seeds.

In addition to the absence of stress or competition, RPM™ seedlings have an extremely dense, fibrous root system with numerous lateral roots providing a greater amount of surface area for the absorption and utilization of oxygen, water, and nutrients than bare-rooted seedlings (Lovelace 1998). Optimal growing conditions provided by the RPM™ system facilitate the production of a dense, fibrous root system and remove the effect of acorn mass, size, and precocious shoot growth on oak seedlings after one growing season.

Table 3.—*The effect of acorn mass on first-year morphology of RPM™ containerized* Quercus *x.* schuettei *seedlings. Values are means ± 1 standard deviation.*

Variables	Heavy	Light	Sign.[2]
RCD[1] (mm)	12.4 ±1.5	10.3 ± 0.4	No
Height (cm)	100.7 ± 8.9	81.0 ± 2.2	Yes
Root volume (ml)	122.9 ± 27.7	88.3 ± 9.4	Yes
Shoot dry weight (g)	38.8 ± 6.3	23.6 ± 3.8	Yes
Root dry weight (g)	44.7 ± 11.6	30.4 ± 4.4	Yes
No. of flushes	4.6 ± 0.5	4.2 ± 0.5	No

[1] Root collar diameter (measured 2.5 cm above the root collar)
[2] Means within a row differ significantly at the 5% level.

Table 4.—*The effect of acorn size on first-year morphology of RPM™ containerized* Quercus bicolor *seedlings. Values are means ± 1 standard deviation.*

Variables	Large	Small	Sign.[2]
RCD[1] (mm)	12.5 ± 1.2	12.6 ± 0.9	No
Height (cm)	103.8 ± 12.1	112.8 ± 13.2	Yes
Root volume (ml)	113.3 ± 27.1	128.3 ± 21.2	Yes
Shoot dry weight (g)	39.1 ± 10.0	42.9 ± 10.7	No
Root dry weight (g)	46.7 ± 11.2	52.6 ± 9.2	No
No. of flushes	4.9 ± 0.4	5.0 ± 0.3	No

[1] Root collar diameter (measured 2.5 cm above the root collar)
[2] Means within a row differ significantly at the 5% level.

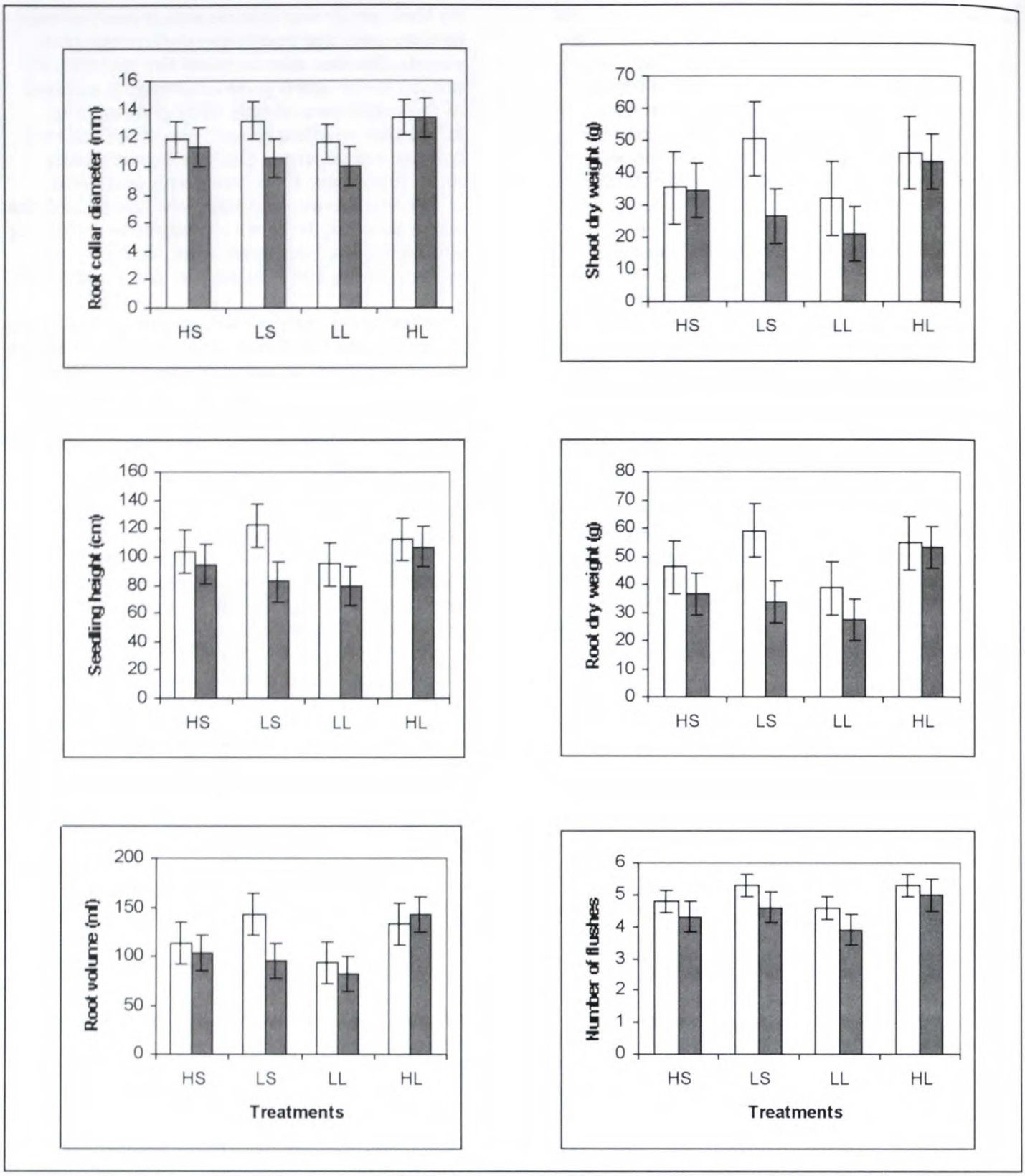

Figure 2.—*The effect of acorn mass and size and early growth on first-year morphology of RPM™ container-grown seedlings from two* Quercus *populations; Treatments: LSN = light-small-non-precocious; LSP = light-small-precocious; LLN = light-large-non-precocious; LLP = light-large-precocious; HSN = heavy-small-non-precocious; HSP = heavy-small-precocious; HLN = heavy-large-non-precocious; HLP = heavy-large-precocious. Means +/- standard error. Open bars=*Quercus bicolor*; shaded =* Q. x. schuettei.

Population Differences

We evaluated the influence of genetics and the RPM™ process on seedling morphology after one growing season by comparing RPM™ seedlings from two different seed sources. Acorns of the hybrid *Q.* x *schuettei* were expected to perform better than the *Q. bicolor* acorns from public collections obtained from the state nursery. However, we found no significant differences in seedling morphology between populations and a few significant differences among treatments. *Quercus bicolor* seedlings were consistently larger than *Q.* x *schuettei* seedlings, but differences were not statistically significant.

There is insufficient evidence to conclude that observed differences in seedling morphology between *Q.* x *schuettei* and *Q. bicolor* acorns after one growing season are indicative of future outplanting success and precocity. Seed collected from superior seed trees have been shown to have increased viability and germinate earlier (Buchschacher and others 1991), growth responses that could prove advantageous in a natural setting (Seiwa 2000). Timing of germination, as well as values of all measured variables, varied more in *Q. bicolor* than in half-sibling *Q.* x *schuettei* acorns, possibly due to the greater degree of genetic diversity found in the randomly collected *Q. bicolor* acorns (Mercier and others 1996).

IMPLICATIONS

Propagation of nursery seedlings requires a large supply of acorns and is often adversely impacted during years of poor seed production. If seedlings were grown under optimal conditions such as in the RPM™ process, it would prove advantageous to retain all viable seed for the production of high quality nursery seedlings. Nurseries using a process similar to RPM™ could produce a greater number of seedlings by eliminating the process of discarding lighter, smaller, and slow-germinating acorns.

A result of retain all viable seed would be that plantings contain a greater amount of genetic diversity, which is advantageous in highly competitive or disturbance prone environments such as floodplains. Variation in germination rates could eliminate complete seedling mortality from late spring floods, where slow-germinating seed would sprout after floodwaters had receded, thus providing some regeneration (Streng and others 1989).

A longer-term objective of this research is to determine if selection for seed size or precocity leads to early and consistent mast production after outplanting. Seedlings not destructively sampled during this study are being outplanted to monitor precocity in mast production.

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DESIGNING AND TESTING A SUBIRRIGATION SYSTEM FOR ROOTING HARDWOOD CUTTINGS

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ABSTRACT.—We explored the feasibility of using a subirrigation system for the propagation of hardwood stem cuttings. Subirrigation systems are relatively inexpensive and simple to construct and maintain. They also offer several advantages for propagating difficult-to-root species by decreasing water usage and nutrient leaching compared to traditional mist propagation systems. Cuttings from six stock types were each randomly assigned to one combination of three cover types, three propagation media, and five auxin treatments. Cuttings under white polyethylene sheeting in coarse perlite had the highest survival percentage and least incidence of basal rot among the subirrigation assemblies. Of the auxin treatments tested, a 3,000-ppm IBA quick dip appears to be a promising auxin treatment for softwood and semi-hardwood cuttings of difficult-to-root hardwoods within a subirrigation system.

The successful propagation of many desirable hardwood trees by means of traditional stem cutting propagation remains an elusive goal. Many species within the Juglandaceae and Fagaceae are highly prized for their timber value and two—black walnut (*Juglans nigra* L.) and pecan (*Carya illinoensis* (Wangenh.) K. Koch)—also produce marketable nuts. Black walnut, pecan, and several species of oak can be important components in several agroforestry practices in the Central Hardwood region (Garrett and Reitveld 1997, Garrett 2003). Clonal propagation of superior trees on their own root systems is highly desired; however, many economically important hardwoods have been classified as difficult-to-root species (Coggeshall and Beineke 1997, Zaczek and others 1997).

Mezitt (1978) first proposed the use of the subirrigation method as a propagation technique for difficult-to-root plants. Holt and Maynard (1997) indicated that a broad range of plants could be successfully propagated—at least to a limited degree—by using subirrigation systems. These systems allow for the upward movement of water from a reservoir through the propagation medium by means of capillary action, rather than from above by means of a standard mist or fog system. The advantages of subirrigation systems include lower cost, reduced water usage and leaching of nutrients, easier maintenance, and avoidance of waterlogged propagation medium (Regan and Henderson 1999).

Various environmental and physiological factors may influence propagation success including composition of propagation medium, amount of light, relative humidity, auxin type and amount, and age and source of cuttings. Difficult-to-root species can be especially sensitive to these factors, which makes their propagation all the more challenging. The purpose of our ongoing study is to develop a simple, inexpensive subirrigation system for cutting propagation of several difficult-to-root tree species.

MATERIALS AND METHODS

Our subirrigation experiments are being conducted in a greenhouse at the University of Missouri's Horticulture and Agroforestry Research Center in New Franklin, MO. Bench light there averaged about 50 percent of full sunlight (PAR) and was supplemented with high-pressure sodium lamps to provide 16-hour photoperiod. Our subirrigation system was

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constructed from two 15.8-inch square by 5-inch tall rigid plastic flats (Anderson Die & Manufacturing Company, Portland, OR). The top tray contained 36 2.4-inch square by 5-inch tall plant bands and was lined with shade cloth to reduce medium loss from the plant bands. The top tray was nested into a bottom or reservoir tray lined with 6-mil clear polyethylene sheeting. The top tray with cuttings was placed on four stacks of small plastic pots that served as 3.5-inch tall pedestals to maintain the proper height of the cutting tray within the reservoir tray with bases of plant bands immersed a maximum of 1.5 inches. Water was added daily to the reservoir tray through an empty plant band to overflow. The reservoir tray was thoroughly flushed weekly.

Frames (2 feet wide x 4 feet long x 2 feet high) were prepared from 1 inch diameter PVC pipe and covered with semi-transparent white 4-mil polyethylene sheeting, 50 percent woven shade cloth, or 80 percent woven shade cloth. Plant bands within each subirrigation assembly were filled with one of the following propagation media: coarse perlite, 9:1 perlite:peat, or 9:1 perlite:crushed limestone. Three assemblies (one for each propagation medium) were randomly placed under each frame openly spaced on a greenhouse bench. Greenhouse temperatures were maintained between 75 and 85 degrees F.

Five different plant sources were used to produce 6- to 9-inch long soft to semi-hardwood cuttings in addition to 3- to 6-inch long softwood cuttings from epicormic sprouts. We began by sticking black walnut cuttings in early June derived from several stock types including seedling stump sprouts or shoots taken from 1-year-old potted grafts. In mid-summer, stump sprouts were again harvested from the same potted seedlings. In early June, we also harvested branch tips from 6-year-old trees of flowering black walnut, bur oak (*Quercus macrocarpa* Michx.), swamp white oak (*Q. bicolor* Willd.), and northern red oak (*Q. rubra* L.). These trees were part of a field trial to evaluate growth and early flowering traits of containerized seedlings produced under the RPM® method (Lovelace 1998). Lastly, softwood cuttings were harvested from epicormic sprouts forced in the greenhouse on the basal branches cut from 5- to 30-year-old walnut, swamp white oak, pecan, honey locust (*Gleditsia triacanthos* L.) and white ash (*Fraxinus americana* L.) (Van Sambeek and others 1998a, 1998b; Van Sambeek 1999).

We tried five different auxin treatments in 50 percent ethanol including 3,000 ppm IBA (15mM IBA), 2,000 ppm IBA and 1,000 ppm NAA by diluting Dip 'N Grow (Astoria-Pacific, Clacka-mas, OR), 2,000 ppm IBA and 1,000 ppm NAA by diluting Wood's Rooting Compound (Earth Science Product Company, Wilsonville, OR), 9,000 ppm IBA, and a 50-percent ethanol control. All cuttings were treated for 5 seconds to a depth of 0.5 inches, and each was randomly assigned to one of the six treatment combinations for propagation medium and covering type. Cuttings were inserted 2 inches deep in the propagation medium.

Dead or diseased cuttings were removed as found and presence/absence of basal rot recorded. Approximately 1 month after sticking, cuttings were evaluated for survival and retention of green foliage. Approximately 2 months after sticking, cuttings were evaluated for presence/absence of basal rot, visible callus, and adventitious roots greater than 0.2 inches long. Cuttings with either callus and/or adventitious roots were transplanted to our greenhouse potting medium and set under intermittent mist. After 1 month, they were moved to a shaded greenhouse bench for an additional month with semi-weekly watering. Cuttings that were still alive after 4 months were analyzed as having rooted.

Presence/absence data for basal rot, 1 month survival, presence of green leaves, and presence of callus for each cutting were converted to treatment percentages by calculating averages for the different treatment combinations of source of cutting, auxin treatment, propagation medium, and type of covering. Number of cuttings ranged from 1 to 7 within any treatment combination. Percentage data were subjected to 4-way ANOVA with 3- and 4-interactions pooled as the error term. All two-way interactions having $p > 0.01$ for a significant F-value were also pooled into the error term in subsequent analyses. Treatment means were separated using Duncan's multiple range test at the $p = 0.05$ probability level. Because so few softwood cuttings from epicormic sprouts survived, they were excluded from further analyses for effects of propagation medium and type of covering.

RESULTS AND DISCUSSION

The source of cuttings had an impact on the percentage of cuttings that developed basal rot, the percentage that were alive 1 month after sticking, and the percentage of live cuttings that retained some green foliage or produced visible

callus (table 1). While the semi-hardwood cuttings from the RPM walnut had the least incidence of basal rot, they also had the lowest percentage of surviving cuttings with callus. In contrast, the softwood cuttings from the RPM oak and the epicormic sprouts had the highest incidence of basal rot and mortality, as well as few cuttings with callus. Teclaw and Isebrands (1987) found successful propagation of oaks to be highly associated with the timing of cutting collection. Our results suggest the need to better assess the physiological age or degree of lignification of our cuttings, especially for the oak species and epicormic sprouts, to reduce fungal disease and mortality.

The relatively high incidence of basal rot may indicate a lack of aeration with our subirrigation system. Rein and others (1991) discussed the need to maintain high oxygen levels in the propagation medium to stimulate rooting. Our system currently does not circulate the water in the bottom reservoir except when the water is replaced weekly by flooding from below through an aerating nozzle. Rein and others (1991) also reported that rooting success is highly dependent upon the growth stage when taking cuttings. We did find a lower incidence of basal rot and a higher percentage of cuttings with callus from the summer stump sprouts compared to spring stump sprouts.

The type of covering influenced the incidence of basal rot, cutting survival, as well as percentage of live cuttings retaining green foliage or producing visible callus (table 2). Cuttings covered with polyethylene sheeting had the lowest incidence of basal rot and highest survival. We also observed that these cuttings had less wilting immediately after sticking, which may account for the higher percentage of cuttings retaining some green foliage a month later. Other studies suggest the retention of leaves on cuttings of difficult-to-root woody plants is important as a source of carbohydrates for plant respiration and root growth (Hartman and others 1997). Zaczek and others (1997) found heavy shade increased rooting of oak cuttings under humidifying fog. Our results indicate maintaining a high humidity under white polyethylene sheeting was more beneficial than keeping cuttings under heavy shade from woven polypropylene fabrics that provided more air movement.

Of the three propagation media, cuttings in coarse perlite had the lowest incidence of basal rot, the highest survival percentage after 1 month, and the highest percentage with callus after 2 months (table 2). The addition of either peat or limestone to the propagation medium failed to improve our results over use of coarse perlite alone. Apparently, the addition of peat or limestone may have resulted in unfavorable pH values, increased the water holding capacity, or decreased oxygen levels in the medium. Both low oxygen and high water holding capacity can lead to increased incidence of fungal infection and basal rot (Loach 1988, Rein and others 1991). Apparently, the physical properties of the propagation medium, such as particle size, are not as critical as the ease with which water can move through it (capillarity) and then be made available to cuttings.

Our results did confirm earlier observations that coarse perlite is an excellent propagation medium (Holt and Maynard 1997, Regan and Henderson 1999, Al-Salem and Karam 2001). Perlite has been the basic medium of choice for most subirrigation systems because it has the high surface to volume ratio needed for aeration and water movement, as well as good thermal insulation characteristics (Cook and Dunsby 1978, Holt and Maynard 1997).

Table 1.—*Effect of stock plant source on percentage of all cuttings with basal rot or surviving for 1 month and on percentage of live cuttings retaining green foliage or producing callus after 2 months in a subirrigation system*

		All cuttings		Live cuttings only	
TREATMENT	Cuttings treated (#)	Basal rot	1 month survival	Green foliage	Visible callus
	- # -	- % -	- % -	- % -	- % -
Forced epicormic sprouts	110	76 a	7 c	——	——
Spring stump sprouts	60	48 b	31 b	76 a	32 b
Summer stump sprouts	156	51 b	43 b	52 ab	52 a
Shoots from grafts	30	39 b	58 a	73 a	47 a
6-year-old RPM walnuts	65	20 c	63 a	29 b	8 c
6-year-old RPM oaks	298	84 a	14 c	18 b	14 c

Means followed by the same letter are not significantly different at the p = 0.05 level according to the Duncan's multiple range test.

Table 2.—*Effect of frame covering and propagation medium on percentage of all cuttings with basal rot or surviving for 1 month and on percentage of live cuttings retaining green foliage or producing callus after 2 months in a subirrigation system*

TREATMENT	All cuttings		Live cuttings only	
	Basal rot	1 month survival	Green foliage	Visible callus
	- % -	- % -	- % -	- % -
TYPE OF COVERING:				
White polyethylene	42 b	52 a	57 a	29 a
50% Shade	60 a	32 b	41 a	42 a
80% Shade	64 a	24 b	36 a	36 a
PROPAGATION MEDIUM:				
Perlite:Peat 9:1	65 a	28 b	41 a	16 b
Perlite	40 b	48 a	53 a	55 a
Perlite:Limestone 9:1	62 a	32 b	45 a	20 b

Means followed by the same letter are not significantly different at the p = 0.05 level (n = 183 to 217) according to the Duncan's multiple range test.

The auxin treatments resulted in a variable response as would be expected with different auxins and carriers (table 3). Cuttings quick-dipped in 3,000 ppm IBA in 50 percent ethanol or the 50-percent ethanol control had the least incidence of basal rot and highest percentage of surviving cuttings. The auxin treatments with the highest incidence of basal rot and fewest surviving cuttings after 1 month were 9,000 ppm IBA or mixtures containing 2,000 ppm of IBA and 1,000 ppm of NAA suggesting these treatments were too strong for our softwood and semi-hardwood cuttings.

Dirr and Heuser (1987) reported that high auxin concentrations could predispose cuttings to basal rot. Mixtures of IBA with NAA (Dip N' Grow and Wood Rooting Compound) were tried because mixtures are frequently more effective than either component alone (Hartman and others 1997, Navarrete and others 1989). No softwood white ash cuttings from epicormic sprouts rooted in our subirrigation system due to the high incidence of basal rot; however, similar cuttings have been rooted under intermittent mist after a quick dip with 600 ppm Dip N' Grow solution (Van Sambeek and others 1998b). The percentage of cuttings with callus or roots was too low for us to evaluate the effectiveness of auxin mixtures with and without the addition of additives such as dimethylformamide in Wood's Rooting Compound. Finally, we were unable to confirm Shreve's (1974) earlier reports that epicormic or stump sprouts of black walnut could be rooted with a quick dip in 8,000 ppm IBA in 50 percent ethanol.

Although more than 700 cuttings either died within 1 month of sticking or failed to initiate roots, we did have one swamp white oak and four walnut cuttings that initiated at least one elongating adventitious root within the first 2 months of sticking. All five cuttings retained some green leaves and rooted in the perlite propagation medium. Two of the four walnut cuttings had been treated with Wood's Rooting Compound, one with 3,000 ppm IBA, and one with 9,000 ppm IBA. The swamp white oak cutting originating from an epicormic sprout had been treated with Wood's Rooting Compound.

Our results confirm previous observations that the most promising subirrigation system for rooting hardwood cuttings will likely use coarse perlite as the propagation medium. We also found that our subirrigation assemblies were relatively inexpensive, could be easily constructed and maintained, and were easily managed if one plant band was left empty to add water to the lower reservoir. We are currently pursuing approaches to increase aeration within the propagation medium to decrease incidence of basal rot. Future work will evaluate the effectiveness of other frame coverings to reduce initial cutting stress. More attention is needed on proper conditioning of stock plants such as use of end-of-day far red light treatments as described by Hawver and Bassuk (2000). Finally, we will continue to evaluate several approaches described by Bassuk and others (1987) to precondition, etiolate, or blanch shoots on stock plants to increase cutting survival or rootability.

Table 3.—*Effect of auxin formulation on percentage of all cuttings with basal rot or surviving for 1 month, on percentage of live cuttings retaining green foliage or producing callus after 2 months in a subirrigation, and percentage of live cuttings successfully transplanted and established under mist*

TREATMENT	Number of cuttings treated	All cuttings		Live cuttings only		
		Basal rot	1 month survival	Green foliage	Visible callus	Rooted cuttings
	- # -	- % -	- % -	- % -	- % -	-%-
50% ethanol	166	49 b	47 a	57 a	41 a	15 a
3,000 ppm IBA	135	34 c	56 a	40 a	26 a	14 a
3,000 ppm Dip N' Grow	135	67 a	22 b	36 a	33 a	5 a
3,000 ppm Wood's RC	138	62 ab	25 b	52 a	33 a	16 a
9,000 ppm IBA	40	75 a	20 b	50 a	42 a	25 a

Means followed by the same letter are not significantly different at the p = 0.05 level according to the Duncan's multiple range test.

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GIANT CANE PROPAGATION TECHNIQUES FOR USE IN RESTORATION OF RIPARIAN FOREST ECOSYSTEMS

Rebecca L. Sexton, James J. Zaczek, John W. Groninger, Stephen D. Fillmore, and Karl W.J. Williard[1]

ABSTRACT.—This study was initiated to determine factors important for rhizome cutting propagation of giant cane for use in canebreak restoration. Experiment I showed that greater numbers of culms were produced for rhizomes with more internodes especially when surface planted rather than buried. Experiment II determined that 76 percent of the 435 rhizomes tested produced at least one culm and, of these, 28 percent had produced two or more culms. Culm production varied by date of collection and collection site (putative genotype). These results suggest that rhizome cutting propagation may be used to improve the success of giant cane restoration efforts.

Giant cane (*Arundinaria gigantea* (Walter) Muhl.), a member of the *Bambusoidae* division of the grass family *Poaceae* (Simon 1986), is a component of bottomland and riparian forest ecosystems in 22 states throughout the southeastern United States. Its range extends from southern Maryland west to southern Ohio, Indiana, Illinois, and Missouri, south to central Florida, and west to Texas (Marsh 1977). Giant cane dominated communities or canebreaks formerly occupied extensive areas throughout the region. However, land conversion for agricultural purposes, grazing, and fire suppression has greatly reduced the number and extensiveness of giant cane dominated communities with most being restricted to small patches. Cane-breaks are now considered by some to be a critically endangered ecosystem (Platt and Brantley 1997, Bell 2000, Platt and others 2001).

The reduction of canebreak habitat has led to a decline in associated wildlife species (Platt and others 2001), most notably Swainson's Warbler (Thomas and others 1996). Giant cane vegetation in riparian zones enhances water quality because of its dense mat of culms and rhizomes that stabilize streambanks. Additionally, riparian zone canebreaks have high infiltration rates and are effective filters of agricultural runoff (Schoonover 2001). Consequently, there is great interest in giant cane restoration within the region. Unfortunately, giant cane restoration has been limited by the lack of available planting stock and the lack of practical methods of propagating the species (Feeback and Luken 1992).

Giant cane can be propagated sexually but seed is sporadically produced and is characterized by low viability (Farrelly 1984, Platt and Brantley 1997). The species also spreads by rhizomes and can be asexually propagated by digging and transplanting culms or culm offsetting. However, this method is labor intensive, cumbersome, and costly (Platt and Brantley 1993). Rhizome cuttings could potentially be used but little is known about factors that are associated with successful shoot (culm) production for the species. The objective of this study was to determine whether giant cane culms can be produced from rhizome cuttings in a framework that produces machine-plantable stock.

METHODS

This paper reports on two experiments. In Experiment I, rhizomes were dug on 2000 September 22 at four different sites in Pulaski County, Illinois. Plant material was bulked together, kept moist, and transported to the Forest Education and Research Station (ForestERS) greenhouses at Southern Illinois

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University for processing. Any attached culms were removed from rhizomes and they were cut into pieces that were 2, 4, or 10 and greater (10+) internodes long. Ninety rhizome sections of each internode length were randomly located on greenhouse benches. To test for differences in planting position, half (45) of the rhizomes of each internode length were placed on the surface of perlite and the other half buried to a depth of 2 cm. Plant material was misted for 12 seconds every 6 minutes during daylight hours.

Production of culms greater than 1 cm long arising from rhizomes was noted through 2000 December 15. Because of malfunctioning mist and heating systems that impacted the study, further work with propagules of Experiment I were discontinued. Comparisons in culm production among rhizomes of varying lengths and planting position were made relative to the number of internodes in each treatment by using chi-square analysis at alpha = 0.05.

Experiment II involved collecting rhizomes from two separate cane patches (putative genotypes, Butter Ridge Road and Hickory Bottoms) in Pulaski County, Illinois, on two dates (2001 February 26 and 2001 March 23). Rhizomes were kept moist and cool until processing at the ForestERS greenhouse within 2 days after digging. Rhizomes with a mean length of 25.9 cm (std. error = 0.25 cm) were planted distal end up slightly off vertical in D40 Deepots (Stuewe and Sons, Inc., Corvallis, OR) having a pot diameter of 6.4 cm by 25 cm deep in pre-moistened peat/composed bark based media. At least 3 cm of each rhizome was not buried in media and exposed to sunlight. After potting, plant material was misted for 12 seconds every 6 minutes during daylight hours. In all, 139 rhizome cuttings from collection date 1 and 296 from date 2 were processed.

On 2001 April 18, the number of culms formed greater than 1 cm long were noted for each rhizome cutting. Rhizomes that formed culms were later transplanted outside to determine future field survival and growth. A chi-square based test procedure was used to test if the mean percentage of rhizomes that produced at least one culm differed by collection date or by collection location (putative genotype) at alpha = 0.05 (Hines and Sauer 1989, Sauer and Williams 1989).

RESULTS EXPERIMENT I

There were 75 culms produced from rhizomes that were surface planted versus 26 culms from those planted below the surface (table 1). The longer rhizome pieces (10+ internodes) tended to produce more culms. This was especially true when left unburied and planted on the surface. When considering only those rhizomes planted below the media surface, culm production was independent of rhizome length treatment when considered relative to the total number of internodes in each rhizome length treatment ($p = 0.200$).

When planted on the surface, culm production was dependent on rhizome length treatment ($p \leq 0.001$). Considering the total number of internodes for each rhizome length (2, 4, and 10+) surface-planted rhizomes with 10+ nodes averaged the fewest number of internodes (7.9) to produce a culm. When multiple new culms were formed on a rhizome, the buds distal to the culm that the rhizome originated from tended to sprout first and these culms were also longer than those of more proximal origin.

RESULTS EXPERIMENT II

Of the 435 rhizomes tested, 76 percent produced at least one culm and, of these, 28 percent had produced two or more culms and 9 percent produced three culms. Of the rhizomes that

Table 1.—*The number of culms produced from buried and surface planted 2, 4, and 10+ internode long sections of giant cane rhizomes cultured under intermittent mist*

Rhizome placement	Number of internodes per section (n = 45 each)	Total internodes	Number of culms produced	Mean number of internodes per culm produced
Buried 2 cm	2	90	0	—
Buried 2 cm	4	180	4	45.0
Buried 2 cm	10+	575	22	26.1
Surface	2	90	4	22.5
Surface	4	180	6	30.0
Surface	10+	513	65	7.9

produced culms, 25 percent produced culms that originated on the portion of the rhizome planted below the potting media surface. Culm production varied by collection site (putative genotype) and date of collection (table 2). Rhizomes from Hickory Bottoms stock were more likely to form at least one culm than those from the Butter Ridge Road site for both the first ($p < 0.001$) and second ($p = 0.005$) collection dates. The percentage of culm producing rhizomes did not differ between dates for collections at Hickory Bottom ($p = 0.590$) but collections from Butter Ridge Road differed among dates ($p < 0.001$).

DISCUSSION

Findings in Experiment I suggested that exposure of rhizomes to sunlight appeared to increase the number of culms that were produced particularly for those that were greater than 10 internodes long. Although not specifically propagating giant cane, Bell (2000) suggests that other leptomorphic (running) bamboo species can be propagated by rhizome cuttings but there is no need for light until culms form. Our observations of rhizomes exposed to light when grown on the mist bench surface or those partially unearthed by digging in the field turn from their normal tan color to green suggesting that they become photosynthetic which may provide needed energy to help stimulate production of culms.

Sudden exposure to light may also stimulate bud break from dormant buds, as is the case in the production of epicormic shoots in trees (Kozlowski and Pallardy 1997). Light stimulation of bud break from dormant buds on rhizomes may have occurred in Experiment II as only 25 percent of the culms were formed below ground even though most of the rhizome length was buried.

Simon (1986) and Bell (2000) suggest that late winter and early spring are recommended times for collecting rhizomes for propagation in related cane species. Our results bear this out as culm production from rhizome cuttings was greater when collected in early spring compared to those collected in late winter (at least for one putative genotype) and especially for those collected in fall.

It has been suggested that rhizome pieces should be from 45-60 cm long for cutting propagation (McClure 1993). In the present study, the rhizomes were about half that size and 76 percent of them produced culms. Smaller rhizomes, if successful at producing culms, offer the advantage of being able to produce more planting stock with less rhizome material and are easier to handle for outplanting.

We found that it is possible to produce giant cane planting stock that is a manageable size for machine planting under field conditions by using rhizome cuttings under intermittent mist in a greenhouse. Preliminary outplanting results suggest that this stock is capable of surviving through at least the first growing season. Utilizing containerized giant cane planting stock may improve the success of canebreak restoration efforts.

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Table 2.—*The influence of date of collection and collection area (putative genotype) on the production of at least one culm from giant cane rhizomes*

Date	Collection area	Number of rhizomes tested	Percentage of rhizomes that produced at least one culm
Feb. 26	Butter Ridge Road	76	60.6
	Hickory Bottom	63	81.0
Mar. 23	Butter Ridge Road	183	77.1
	Hickory Bottom	113	82.1

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NUTRITION AND IRRIGATION REGIME AFFECT SIZE AND EARLY GROWTH OF WHITE OAK SEEDLINGS

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ABSTRACT.—Modifications of our nursery protocol for oaks during 1997 and 1998 by instituting dormant season irrigation resulted in high white oak seedling quality. The improved growth was associated with the consistent presence of succulent fibrous roots on the upper 10 to 15 cm of taproots. Many of these fibrous roots on 1-1 stock were as sturdy as many of the permanent first-order lateral roots. White oak seedlings from the 1998 nursery crop grown under the modified irrigation schedule have developed well in the field. At age 4 many are 4.0 m tall with up to 50 mm stem diameters.

White oak, (*Quercus alba* L., WO), one of the most valuable and versatile oak species throughout its range, is capable of regenerating by either seeds or sprouts (Rogers 1990). The species has a reputation of being a slow grower, and it persists and thrives on low quality sites where few faster growing competitors exist. On high quality mesic sites, its slow growth is often detrimental because it cannot compete with the many faster growing or shade tolerant species. In this situation, artificial regeneration has been attempted to supplement poor naturally regenerated WO seedlings. However, it has met with little success (Rogers 1990).

In the early 1980s, the major impediment to artificial regeneration of WO was the absence of quality seedlings. Some agencies suggested that because artificial regeneration results were so poor, further attempts should be discouraged until nursery and outplanting technology were improved (Boyette 1980, Hill 1986). In 1984, research was initiated at the USDA Forest Service Institute of Tree Root Biology (ITRB) in Athens, Georgia, to develop a nursery protocol for growing oak seedlings that would enable the formulation of a biologically based grading system.

Initially, WO and northern red oak, (*Quercus rubra* L., NRO) were the species receiving the most attention for developing nursery production standards. In 1984, few forest tree nurseries were growing hardwoods in the southern United States because the primary emphasis then was on pine seedling production. However, the oak decline problem was of great concern to groups such as furniture manufacturers and wildlife organizations who were aware of declining oak populations on high quality mesic sites. A potential solution to reverse oak decline led to enrichment plantings in recently harvested or in under-stocked oak decline stands. Unfortunately, this was not vigorously pursued because of WO's reputation for slow growth in the nursery. Average heights (HGT) of shippable seedlings were less than 25 cm tall with root collar diameter (RCD) of 4 mm or less (Wichman and Coggeshall 1983).

This paper does not report the results of a single study, but rather summarizes a gradually evolving procedure for producing seedlings with consistent morphology conducive for artificial regeneration of WO. Until we lifted and evaluated the 1996 nursery crop, WO root system morphology was considered to be different from other oaks tested. White oak had consistently produced fewer first-order-lateral roots (FOLR) and fibrous feeder roots along the first 10 to 15 cm of the taproot.

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In all our previous nursery trials, WO seedlings never approached the stem sizes attained by NRO seedlings. The initial nursery trials with WO at the ITRB experimental nursery used the nursery seedling sizes obtained from the Georgia Forestry Commission's Nursery Program as "standards." The seedlings produced in our initial trials did not represent a significant improvement in quality from those produced in the Georgia Forestry Commission or reported by others (Wichman and Coggeshall 1983, Rogers 1990).

NURSERY TRIALS BEFORE 1990

In 1986, after several modifications in seasonal N application rates, a significant improvement in WO seedling sizes was finally realized (Kormanik and others 1989). Afterwards, we did little research on WO for several years due to poor acorn availability. Rather, we concentrated on developing nursery protocols for NRO for which acorns were readily available from the USDA Forest Service Watauga Seed Orchard located in eastern Tennessee. After developing a reliable protocol for NRO nursery production in the early 1990s, we resumed research on improving WO seedling morphology (Kormanik and others 1994).

The initial nursery research prior to 1990s was done with acorns collected from grafted mother trees established at the USDA Forest Service Beech Creek Seed Orchard in western North Carolina, using the same nursery fertility protocol that was developed for NRO (Kormanik and others 1995, 1998). Acorn germination from these grafted mother trees ranged between 10 to 60 percent despite extra care that was used for collection and storage.

Although poor germination resulted in abundant nursery bed spacing, these WO acorns produced seedlings of unsatisfactory quality. We continued to test the Beech Creek Seed Orchard WO almost exclusively since they represented years of orchard development by the USDA Forest Service Southern Region genetics program and contained mother trees from throughout the southern Appalachian Mountain range.

Data from almost all these grafted selections were poor and difficult to interpret. For example, it was common to have a significant percentage (10 to 20 percent) of the acorns germinate but never develop an epicotyl. In these cases, the radicle would develop normally until the stored food in the cotyledons was exhausted and then it would begin to deteriorate. The cause of this abnormality is unknown. Absence of epicotyl development is not common, but we could find no evidence of insect damage, epicotyl breakage, or fungal damage to account for failure of epicotyls in our trials.

Equally disturbing was the percentage of the developing seedlings that had multiple tops and other maladies that were considered unacceptable for outplanting stock. Eventually, WO acorns from wild collections were included in our tests along with those from the Beech Creek grafted orchard. Poor germination and other abnormalities so common to the WO seedlings from grafted stock acorns were seldom observed with seedlings obtained from wild acorn collections.

NURSERY TRIALS 1990 TO 1996

Early tests using acorns from non-grafted parents germinated well and produced seedlings that developed sizes comparable to those reported earlier (Kormanik and others 1989). However, we continued to find that less than 40 percent of the WO seedlings were of acceptable quality for outplanting. These seedlings characteristically had few FOLR and fine feeder roots along the top 10 to 15 cm of the taproot.

Our root evaluation system, which relied heavily on numbers of permanent FOLR that developed along the top 30 cm of the taproot, yielded a distribution of 30, 30, and 40 percent for good, intermediate, and cull seedlings, respectively. Variations in seedling root morphological development and its relationship to seedling size proved perplexing. Larger seedlings were obviously associated with better FOLR root development, but the absence of FOLR was not always associated with undesirable stem development.

In 1996, a greenhouse test was conducted to study early taproot development following fall sowing in an attempt to mimic early seasonal growth in outdoor nursery beds. Usually, there is a long period between rainfalls during the post-fall acorn sowing period and epicotyl emergence in early spring. Irrigation prior to epicotyl emergence was not considered important in outdoor nursery beds and is still seldom practiced in southern nurseries.

Irrigation of the acorns germinating in the greenhouse was routine and was always considered essential for their development. This research showed that the taproots from the greenhouse had abundant feeder root development before epicotyl emergence and subsequent development of more permanent FOLR. This

early dormant season development of the WO radicle is well known, but past research showed that comparable feeder root development was not always occurring in our nursery beds. Early presence of such feeder root development in late winter or spring in our outdoor trials had not been of concern, and no attempt had been made to follow their development.

CURRENT NURSERY PROCEDURES

The baseline fertility level developed by the USDA Forest Service's ITRB in conjunction with the Georgia Forestry Commission is used to produce oak seedlings and 35 other hardwood species. All fields in the nursery are tested annually for nutrient status and the soils are adjusted to correct any significant departures from the baseline protocol. The soil nutrient concentrations at the Flint Nursery are adjusted to maintain extractable levels of Ca, K, P, Mg, Cu, Zn, and B at 500, 130, 100, 50, 0.3 to 3, 3 to 8 and 0.4 to 1.2 parts per million, respectively.

For oaks and most other hardwood species, 1,345 kg ha^{-1} of NH_4NO_3 is applied to the developing seedlings throughout the growing season. Small initial applications are gradually increased as the seedlings develop and utilize the short-lived nitrogen applications. Thus, the first two applications are generally at rates equivalent to 17 kg ha^{-1} and the third at 56 kg ha^{-1}. The next six applications are at 168 kg ha^{-1} and the final two at 112 kg ha^{-1}. Nitrogen applications begin in mid-May and continue at 10-day intervals until mid-September. Nitrogen top dressing is reduced or even discontinued in early fall to stop height growth when seedlings are more than 1 m tall. Irrigation is applied as needed to prevent water stress or to reduce heat stress during the growing season.

By 1998, it became apparent that our basic fertility protocol was just as effective with WO as it has been with NRO at our experimental nursery and at the Georgia Forestry Commission's Flint River Nursery as long as irrigation was applied periodically during the dormant season to prevent desiccation of the acorns and developing radicles. Dormant season irrigation of nursery beds, coupled with the ITRB basic fertility baseline protocol, has resulted in over 50 percent of WO seedlings now either meeting or exceeding minimum standards originally set up for the faster growing NRO seedlings. Those standards for FOLR number, HGT, and RCD are: 6, 70 cm, and 8 mm, respectively.

RESULTS

Improvement in WO seedling quality has been steady since the realization of the importance that dormant season irrigation has on WO root morphology. The 1986 data in figure 1 represented what we thought were acceptable distributions and sizes for WO seedlings under the nursery management protocol at that time (Kormanik and others 1989). The seedlings from the 1986 data still essentially represent WO seedling production standards in the southern United States for 1-0 stock. The data from the 1998 and 2000 indicate the potential improvement easily attained by using the modified ITRB nursery protocol.

Nursery Trials

In 2001, a portion of the 2000 experimental nursery crop was replanted into the beds after being evaluated for size and FOLR number.

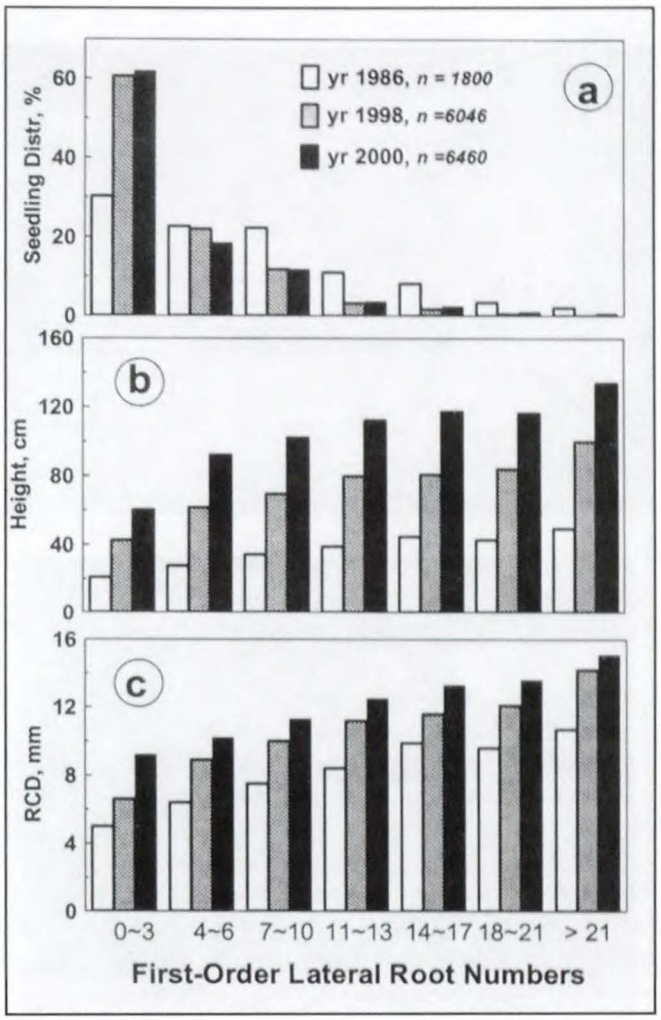

Figure 1.—*Mean values for (a) seedling distribution by first-order lateral root (FOLR) groups, (b) height, and (c) root collar diameter (RCD) for 1-0 white oak seedlings during three different years of production with an Institute of Tree Root Biology nursery protocol.*

Until the 1-1 stock was lifted from the nursery beds and re-evaluated in 2002, the FOLR group (0-3) was generally considered to be culls because of their substandard sizes (figs. 1b and 1c). Many of the seedlings in the 4-6 FOLR category were also placed in the cull grouping because they were below minimum shippable standards for both HGT and RCD. Past experiences showed that few of these individuals in the 0-3 or 4-6 FOLR groupings proved vigorous when transplanted into fertile nursery beds for further development as 1-1 stock. Indeed, most seedlings that were not competitive as 1-0 stock failed to meet standards the second year, even in fertile, irrigated nursery transplant beds (fig. 2).

In 2000, based upon observations of root development following dormant season irrigation, an additional criterion was added to our seedling evaluation procedure for WO: the presence of succulent fibrous roots along a seedling's taproot. This condition was rarely encountered before dormant season irrigation was initiated. Thus, its potential importance was overlooked. Previously, seedlings with zero FOLR were always classified as "cull", and were seldom deemed to be of shippable quality.

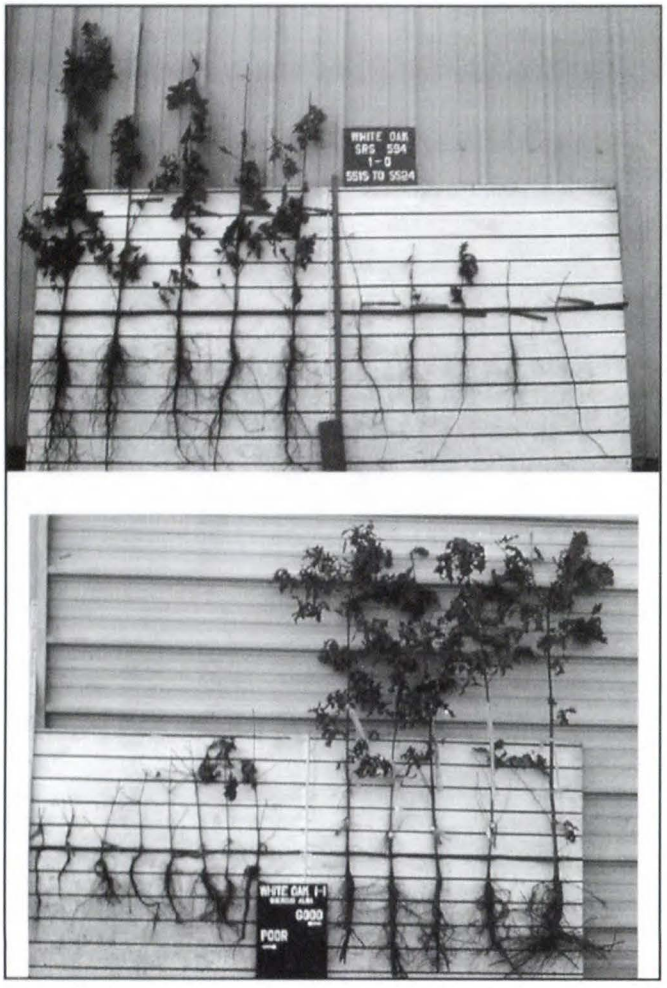

Figure 2.—*The seedlings in the best and poorest 25 to 30 percent of any half-sib progeny groups are significantly different in sizes and competitive ability. (a) White oak seedling as 1-0 stock, (b) white oak transplanted into nursery beds and lifted as 1-1 stock.*

In the year 2000 crop, this zero FOLR group represented 30 percent of the 1-0 seedling crop (figs. 2 and 3a). In addition, many seedlings in the 1-3 FOLR group were also down graded to cull because of their small sizes, resulting in an additional 33 percent to the cull group (fig. 3). This 63 percent total was comparable to the cull percentage that had remained stable with our nursery protocol since the 1986 crop (Kormanik and others 1989). The 35 to 40 percent of WO seedlings in the ≥ 4 category was also a rather consistent percentage and characteristically represented shippable seedlings (fig. 3a).

When the graded seedlings were lifted as 1-1 stock, the value of the succulent fibrous root systems became readily apparent (figs. 3b, 3c, and 3d). Those 1-0 individuals with 3 or fewer FOLR and no succulent fibrous roots (21 and 14 percent) generally developed the fewest new FOLR and had the poorest stem development (figs. 3c and 3d). Compared with 1-0 seedlings with an abundance of succulent, fibrous roots, 1-0 stock with fewer than four FOLR and without abundant fibrous roots grew fewer new FOLR and much less in HGT and RCD during the additional year in nursery beds. In all cases, those seedlings with FOLR of ≥ 4 and with or without fibrous roots yielded the best development as 1-1 stock (figs. 2, 3c, and 3d).

The importance of dormant season irrigation to the morphological development of WO seedling roots and subsequent seedling vigor must be stressed. Dormant season irrigation is a new development and is not yet commonly practiced in forest tree nurseries in the southern United States where soils do not freeze during the winter period. Thus, we now consider for inclusion in "cull" only those seedlings without either adequate FOLR or succulent fibrous roots.

For the past 3 years, dormant season irrigation has essentially reduced our culling rate from about 65 percent to 30 percent in our research trials and in the commercial production of the Georgia Forestry Commission nursery. The best 35 to 40 percent of the seedlings have consistently performed well after outplanting in the field. Those from the lowest 30 percent have not

performed adequately even when transplanted into fertile nursery beds.

White Oak Field Performance

The Georgia Forestry Commission began developing new WO seedling seed orchards, based primarily on seedlings with the most robust root system. Of course, these also were the tallest seedlings with the largest RCD. In 1994, using FOLR number as the strongest grading criterion, a second-generation seed orchard was established adjacent to their Flint River Nursery. By ages 5 to 6, many of these trees were producing acorns in sufficient numbers that allowed progeny testing to begin in the nursery. In year 2000, a third generation WO seed orchard was developed from a 6-year-old second-generation seed orchard (Georgia Forestry Commission 2000).

In 1998, the first outplanting of WO seedlings was undertaken using seedlings that had benefit of dormant season irrigation. This potential seed orchard established in a small clearcut in North Georgia includes 30 different progeny groups with a total of 1,250 seedlings. Without supplemental irrigation or fertilization, most family progenies have 95 percent survival and many trees were 3.5 to 4.3 m tall with DBH of ≥ 50 mm at age 4. The only exceptions are those seedlings from grafted mother trees. Their survival is poor, with 100 percent mortality in some plots. This performance is even more impressive in the light of record-setting droughts endured by these seedlings since their establishment.

CONCLUSIONS

In the southern states where it is best to sow oak during late fall, irrigation is essential for preventing acorn desiccation prior to radicle emergence and for providing conditions for

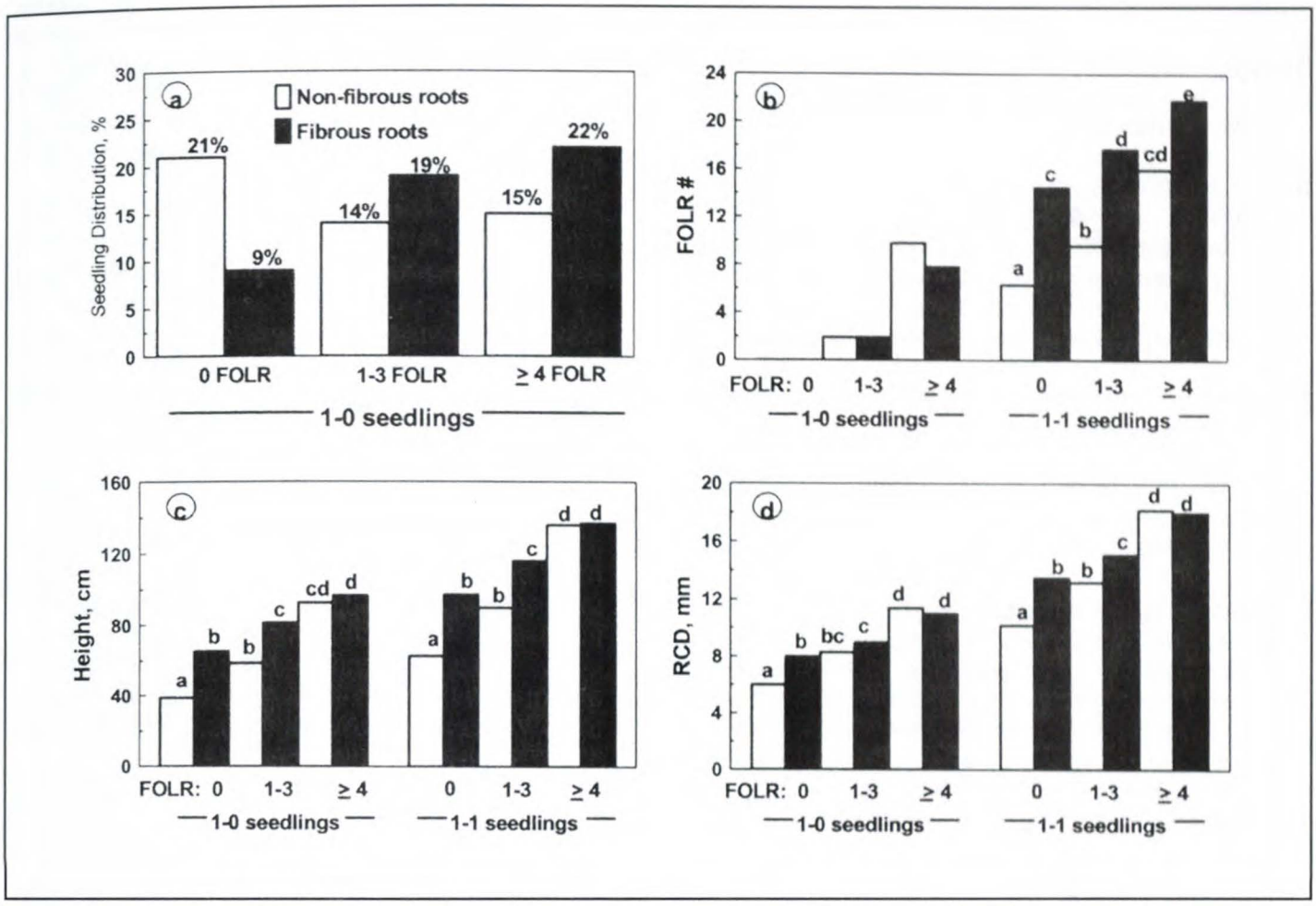

Figure 3.—*Mean values for (a) seedling distribution by specific first-order lateral root (FOLR) groups and presence of fibrous roots, (b) FOLR number, (c) height, and (d) root collar diameter (RCD) for 1-0 white oak seedlings grown in 2000 and then transplanted into similar nursery beds and grown for another year. Data presented here are from seedlings which were systematically selected as every tenth seedling in nursery beds during 2000. Means with same letters within an age class are not significantly different based on Tukey's test at the P = 0.05 level.*

optimum taproot development prior to epicotyl emergence. The nursery fertility protocol developed by the USDA Forest Services ITRB has consistently produced large competitive NRO, but until dormant season irrigation was initiated, WO seedlings had much smaller size.

Dormant season irrigation has increased shippable seedlings from 40 to 70 percent for some families' progeny. The small seedlings representing 30 percent cull group have seldom been competitive, either in nursery transplant beds or under field conditions. However, those from the best 40 percent have normally made satisfactory growth in the field. Those from the middle 30 percent may need more maintenance to remain competitive than normally available.

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PLANTED AND NATURAL TREE SEEDLING SURVIVAL AND DENSITY IN THREE FLOODPLAIN RESTORATIONS ON ABANDONED AGRICULTURAL FIELDS

Allen E. Plocher[1]

ABSTRACT.—In three floodplain forest restorations, established in abandoned agricultural fields in Illinois, permanent plots were sampled for 3 years to determine survivorship and density of planted tree seedlings, and species composition and density of natural regeneration. Planted tree survivorship decreased over time at all sites and after 3 years ranged from 32 to 61 percent. Planted density ranged from 780 to 1,330 individuals/ha and did not differ between species. Sapling/shrub stage natural regeneration ranged from 133 to 7,017 individuals/ha. Natural regeneration equaled or surpassed planted seedling density at two of the three sites. The site with greatest natural regeneration also had highest survival and density of planted trees.

Bottomland forests in the midwest have experienced extensive reduction and degradation since the 1930s. Large areas have been cleared for soybean production (Newling 1990). By the end of the 1980s, 81 percent of bottomland forest area had been lost (Allen and Kennedy 1989, Clewell and Lea 1990). Efforts to restore these systems only began in the 1980s and there is, so far, little published work on restoration of forested wetlands (Clewell and Lea 1990, Newling 1990).

Secondary succession in old fields in bottomlands proceeds at a rapid rate as long as propagules are reasonably nearby. Rapid site invasion is seen in tree species with wind dispersed seeds and in shrubs with seeds dispersed by water or animals. Heavy seeded species such as oaks, hickories, and pecans, invade much more slowly, making reintroduction of these species a major concern in restoration (Newling 1990).

A critical gap in research on bottomland hardwood forest regeneration is the identification of favorable conditions for natural regeneration and survival of planted wetland tree species. Studies show that with a seed source nearby natural regeneration density and survival may be equal to or greater than that of planted seedlings (Clewell and Lea 1990, Kolka and others 1998). In upland areas, hardwood plantations are usually unsuccessful without such weed control practices as herbiciding, mowing between rows, and installing tree cones (Allen and Kennedy 1989, Hansen and others 1993). In bottomlands, however, tall herbaceous weeds may be beneficial to planted seedlings by providing shelter from high light levels and temperatures, wind, and herbivores (Clewell and Lea 1990, McLeod and others 2000).

Woody cover may be beneficial by providing shade and producing humus (Clewell and Lea 1990). It is uncertain when a bottomland tree planting reaches the point where survival to maturity is likely. Natural tree regeneration experiences extremely heavy mortality for the first 5 years (Harper 1977). In these bottomland systems, chances for success have been found to be excellent with 35 percent survival or 1,161 stems/ha after 2 years (Newling 1990), or with at least 988 stems/ha when an average height of 2 m is attained (Clewell and Lea 1990).

The purpose of this study is to examine survival and density of planted bottomland tree species and density of natural shrub/sapling stage regeneration after 3 years at three sites in Illinois. We determine whether significant

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planted seedling mortality occurred between the second and third years and compare survival rates between four planted species. Planted tree density is compared with that of natural shrub/sapling stage regeneration.

STUDY AREA

The three study sites are in Illinois, along a north-south gradient of 443 km (fig. 1). The Rock River site is located adjacent to the Rock River in Henry County, IL (41° 33′ N, 90° 11′ W), near the town of Joslin. The Illinois River site is located 6.8 km from the Illinois River in Cass County, IL (40° 04′ N, 90° 18′ W) in the extensive lowlands surrounding the Sangamon River's confluence with the Illinois (the Sanga-nois). The nearest town is Beardstown. The Bankston Fork site is located 149 m from Bankston Fork in Saline County, IL (37° 44′ N, 88° 42′ W), near the town of Carrier Mills.

The Rock River and Illinois River sites occur on floodplains, while the Bankston Fork site occurs on glacial lake plain (Willman and Frey 1970). The Rock and Illinois are large streams, approximately 190 m in width at the study sites (Rock River 57 year mean discharge – 6,424 ft^3/sec, Illinois River 10 year mean discharge – 19,464 ft^3/sec) (Sullivan and others 1990, Wicker and others 1998). Bankston Fork is a small stream, less than 10 m wide.

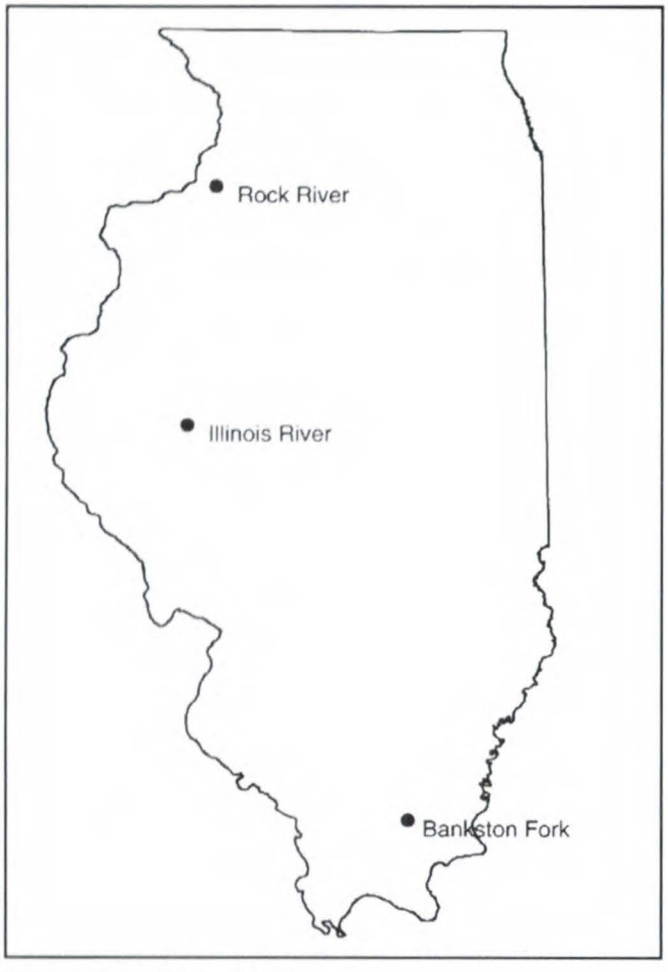

Figure 1.—*Study areas.*

Soils at the three sites are similar and range from silt loam to silty clay loam, and from Typic Fluvaquent to Cumulic Endoaquoll. The Rock and Illinois River sites generally flood during the winter and early spring, and again once or twice early in the growing season (Fucciolo and others 2000, Sanganois State Wildlife Management Area staff personal communication). The Bankston Fork site rarely, if ever, floods but experiences saturated soil conditions early in the growing season (Fucciolo and others 2000).

All three sites are abandoned cropland. The Rock and Illinois sites were in soybean 2 years prior to planting. The Bankston Fork site was cropped approximately 10 years before planting. All sites are adjacent to bottomland forest 50 to 60 years of age.

METHODS

Sites were cleared of vegetation and planted with 2-0 tree seedlings. Planting densities were similar: Rock River at 2,485 stems/ha, Illinois River at 2,149 stems/ha, and Bankston Fork at 2,470 stems/ha. Planting was carried out in the spring of 1998. At Illinois River and Bankston Fork, sites were simply left to revegetate naturally after planting. The Illinois River site is 1.7 ha in size and Bankston Fork is 0.8 ha.

At Rock River the site was more intensively managed. Planting cones were placed around seedlings, and the area between the rows was mowed and herbicided annually. Natural regeneration was allowed to proceed in an adjacent area. The planted area is 2.2 ha and natural regeneration occupies an adjacent 1.2 ha.

Seedlings planted were bottomland species native to the region at each site. Six species were planted at the Rock River site, four species at Illinois River and nine species at Bankston Fork (table 1). Green ash (*Fraxinus pennsylvanica* Marsh), pin oak (*Quercus palustris* Muenchh.), swamp white oak (*Q. bicolor* Willd.), and pecan (*Carya illinoensis* (Wang.) K. Koch) were planted at all sites. Other species planted at one or two sites include overcup oak (*Q. lyrata* Walt.), river birch (*Betula nigra* L.), red maple (*Acer rubrum* L.), silver maple (*A. saccharinum* L.), cottonwood (*Populus deltoides* Bartr. ex Marsh.), sweetgum (*Liquidambar styraciflua* L.),

and black tupelo (*Nyssa sylvatica* Marsh.). At each site, all species were planted in equal numbers. Other tree species naturally occurring at one or more sites included American elm (*Ulmus americana* L.), sycamore (*Platanus occidentalis* L.), peachleaf willow (*Salix amygdaloides* Anderss.), black willow (*S. nigra* Marsh.), catalpa (*Catalpa speciosa* Warder), hackberry (*Celtis occidentalis* L.), white mulberry (*Morus alba* L.), and honeylocust (*Gleditsia triacanthos* L.).

At the Rock River and Bankston Fork sites, shallow hydrologic monitoring wells, stage gauges and water level and precipitation data loggers were installed and monitored monthly by Illinois State Geological Survey scientists (Fucciolo and others 2000). At all three sites, Illinois Natural History Survey soil scientists examined cores and mapped and described the soils.

Vegetation was quantified using stratified sampling. For the understory (tree seedlings < 1 m tall and herbaceous species), parallel transects were established perpendicular to the long axis of the sites at 15 m intervals at Bankston Fork and 20 m at Rock River. Understory vegetation was sampled as aerial cover (and number of stems for tree seedlings) by species in 1 m^2 quadrats at 15 m intervals along transects. At Rock River, 40 quadrats were sampled, and 10 quadrats at Bankston Fork. Understory vegetation was not sampled quantitatively at Illinois River; species dominance was determined by visual estimation.

Importance Values (IV's) were calculated as (relative frequency + relative cover)/2. Dominant species are considered to be those with IV's greater than 6 that are included when, starting with the most abundant, IV's are summed until a value of 50 is exceeded (modified from Federal Interagency Committee for Identifying and Delineating Jurisdictional Wetlands 1989).

For planted trees and sapling/shrub stage (≥ 1 m tall and < 10 cm DBH) natural regeneration at Illinois River, parallel transects were established at 50 m intervals with sample points every 15 m along transects. At Bankston Fork and for natural regeneration at Rock River, transects were established at 30 m intervals with sample points every 30 m. Sapling/shrub stage vegetation was tallied as number of individuals by species, planted or naturally occurring, in 100 m^2 plots. At Rock River, since planted trees occurred in well-defined and maintained rows, 30 m row sections were sampled at 300 m intervals.

Importance Values (IV's) were calculated as (relative frequency + relative density)/2. At Illinois River 14 plots were sampled and 6 plots at Bankston Fork. At Rock River, 14 planted tree plots and 15 natural regeneration plots were sampled. Plots were sampled annually at the end of the growing season. Nomenclature is according to Mohlenbrock (1986).

For statistical analyses, assumptions of normalcy were addressed using Kolmogorov-Smirnoff tests and normal probability plots. Independence of error was evaluated by examining plots of residual vs. predicted values. Data for the

Table 1.—*Planted and naturally occurring tree species (see methods for full scientific and common names)*

BANKSTON FORK	
Planted	**Natural**
Frax. pennsylvanica	*Frax. pennsylvanica*
Quer. palustris	*Quer. palustris*
Quer. bicolor	*Acer rubrum*
Carya illinoensis	*Ulmu. americana*
Quer. lyrata	*Plat. occidentalis*
Betu. nigra	*Popu. deltoides*
Acer rubrum	*Sali. amygdaloides*
Liqu. styraciflua	*Betu. nigra*
Nyssa sylvatica	*Cata. speciosa*
ILLINOIS RIVER	
Planted	**Natural**
Frax. pennsylvanica	*Frax. pennsylvanica*
Quer. palustris	*Acer saccharinum*
Quer. bicolor	*Acer negundo*
Carya illinoensis	*Ulmus americana*
	Plat. occidentalis
	Popu. deltoides
	Salix nigra
	Salix amygdaloides
	Celt. occidentalis
	Morus alba
	Gled. triacanthos
ROCK RIVER	
Planted	**Natural**
Frax. pennsylvanica	*Frax. pennsylvanica*
Quer. palustris	*Acer saccharinum*
Quer. bicolor	*Ulmus americana*
Carya illinoensis	*Popu. deltoides*
Acer saccharinum	*Salix nigra*
Popu. deltoides	*Salix amygdaloides*
	Morus alba

three sites were analyzed separately. At Illinois River, n=14. At Bankston Fork, n=6. For planted trees at Rock River, n=14 and for natural regeneration, n=15. Alpha levels of 0.05 were used to determine statistical significance.

In order to study change in planted tree survivorship (density) over time, one-tailed, unpaired T-tests were conducted to determine whether there were decreases between consecutive years (2000-2001). To determine whether there were differences between species in planted tree survivorship, one-way ANOVAs were conducted using species as treatments and densities (2001) as dependent variables. The analyses were limited to the four species present at all sites: green ash, pin oak, swamp white oak, and pecan.

Two-tailed T-tests were conducted to determine whether, after 3 years, there were differences between planted and natural tree densities. Because planted and natural stems occurred in the same plots at Illinois River and Bankston Fork, paired tests were conducted. An unpaired test was used at Rock River because planted and natural stems occurred in different areas. Because significant results for Kolmogorov-Smirnoff tests and presence of skewness in some data sets suggested problems with normalcy, and plots of residual vs. predicted values showed presence of outliers in some cases, a nonparametric Kruskal-Wallis test was conducted along with each parametric test. Test results were in agreement in every case (Systat, Inc., Evanston, IL).

RESULTS

Within the first year, all three sites supported dense herbaceous communities greater than 1 m in height (table 2). The Bankston Fork herb layer was dominated by inland rush (*Juncus interior* Wieg.), slender mountain mint (*Pycnanthemum tenuifolium* Schrad.), late flowering thoroughwort (*Eupatorium serotinum* Michx.), and Canada goldenrod (*Solidago canadensis* L.). Reed canary grass (*Phalaris arundinacea* L.), bur cucumber (*Sicyos angulatus* L.), giant smartweed (*Polygonum pensylvanicum* L.), and Ontario aster (*Aster ontarionis* Wieg.) were dominant at Rock River. Panicled aster (*Aster simplex* Willd.) and late flowering thoroughwort were the dominant herb layer species at Illinois River.

After 3 years, sapling/shrub stage natural regeneration ranged from 133 stems/ha at Rock River to 7,017 stems/ha at Illinois River (table 2). Seedling stage regeneration ranged from 5,750/ha at Rock River to 53,397/ha at Bankston Fork. Black willow and cottonwood dominated the sapling/shrub layer at Rock River while silver maple and cottonwood dominated at Illinois River. At Bankston Fork, green ash and red maple were dominant. Green ash was the most abundant tree seedling at Illinois River and Bankston Fork while silver maple was most numerous at Rock River.

In 2001, at the end of 3 years, planted tree density ranged from 780 stems/ha at Bankston Fork to 1,330/ha at Illinois River (fig. 2). Planted tree survivorship after 3 years was 61.9 percent at Illinois River, 43.5 at Rock River, and

Table 2.—*Dominant understory species, natural tree seedling density and shrub/sapling layer density as either importance value or number of individuals/ha*

BANKSTON FORK		ILLINOIS RIVER		ROCK RIVER	
Dominant Understory sp.	**IV**		**Estimate**		**IV**
Juncus interior	12.3	*Aster simplex*	dominant	*Phalaris arundinacea*	28.7
Pycnanthemum tenuifolium	10.0	*Eupatorium serotinum*	dominant	*Sicyos angulatus*	12.0
Eupatorium serotinum	7.7			*Popu. pensylvanicum*	9.3
Solidago canadensis	6.8			*Aster ontarionis*	9.1
Fraxinus pennsylvanica	6.5				
Dominant Tree Seedlings	**no./ha**		**Estimate**		**no./ha**
Fraxinus pennsylvanica	25,599	*Fraxinus pennsylvanica*		*Acer saccharinum*	3,750
Ulmus americana	19,997	*Acer saccharinum*		*Ulmus americana*	750
Acer rubrum	7,801	*Ulmus americana*		*Morus alba*	750
Total	53,397	Total		Total	5,750
Dominant Sapling/Shrubs	**no./ha**		**no./ha**		**no./ha**
Fraxinus pennsylvanica	300	*Acer saccharinum*	2,627	*Salix nigra*	60
Acer rubrum	240	*Populus deltoides*	1,977	*Populus deltoides*	60
Total	1,000	Total	7,017	Total	133

31.6 percent at Bankston Fork (fig. 3). Planted tree density was found to decrease between the second and third year (2000-2001) at the Rock River site (14.9 to 10.8 individuals/100 m^2), but did not change at Illinois River or Bankston Fork (fig. 2).

Although green ash had the greatest planted tree density after 3 years at Rock River and Bankston Fork, test results on planted tree density by species were not significant for any site (fig 4). After 3 years, natural sapling/shrub density was greater than planted tree density at Illinois River (70.2 vs. 13.1 individuals/100 m^2) and less than planted tree density at Rock River (1.3 vs 10.8 individuals/100 m^2). Natural sapling/shrub density and planted tree density did not differ at Bankston Fork. Illinois River, the site with the highest survival rate and density of planted trees, also had the greatest density of natural shrub/sapling stage regeneration (fig. 5).

DISCUSSION

Because 35 percent survival (Newling 1990) and 400 stems/acre (Clewell and Lea 1990) after 2 or 3 years have been reported as promising results, good to at least adequate planted tree survival and density was observed at all sites (32 to 61 percent, 316 to 538 stems/acre). In three other studies of bottomland tree restorations, third year survival ranged from 58 to 84 percent (Kolka and others 1998, Dulohery and others 2000, McLeod and others 2001), and a fourth study reported 10 percent survival after 5 years (Conner and others 2000).

Two of the three sites (Illinois River and Bankston Fork) in the present study exhibited no significant mortality between the second and third years. This may indicate that the high mortality phase associated with initial seedling establishment is coming to an end, and provides further evidence that 3 to 5 years after planting is a good time to judge chances for success.

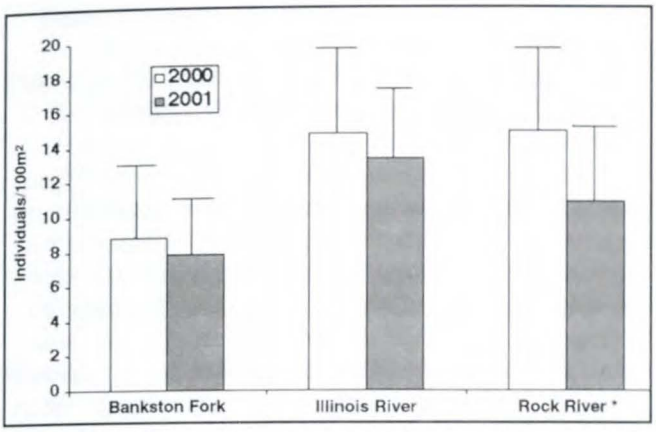

Figure 2.—*Planted tree mortality (mean, standard dev. of decrease in density), 2000-2001.*
** = significant decrease.*

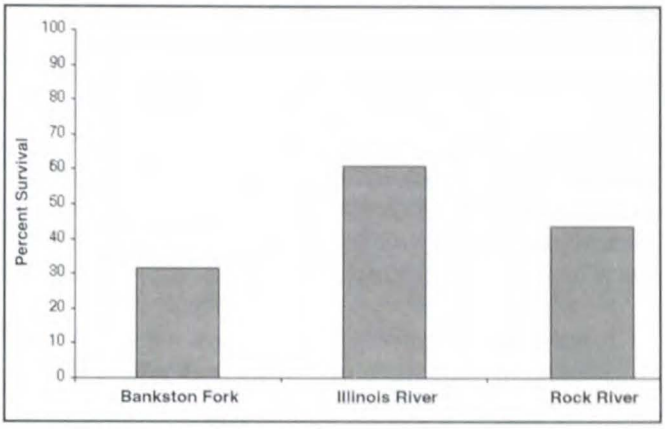

Figure 3.—*Planted tree survivorship (percent survival) after 3 years.*

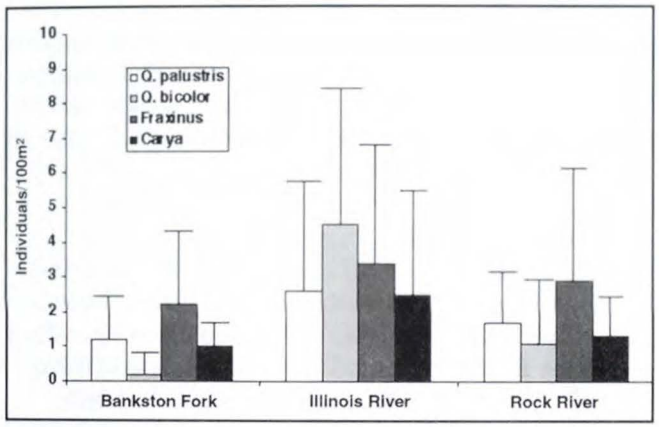

Figure 4.—*Planted tree survivorship (mean, standard dev. of density) by species after 3 years.*

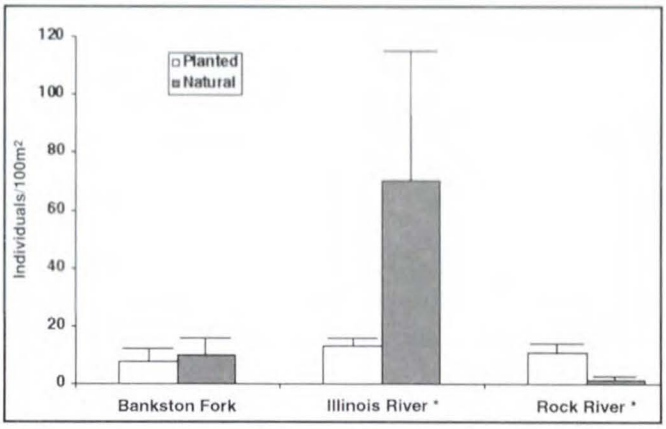

Figure 5.—*Planted tree density vs. natural sapling/shrub density (mean, standard dev.) after 3 years.*
** = significant difference.*

The lack of difference in survival between species at all sites was surprising. At Rock River and Bankston Fork, green ash density was almost twice the average planted tree density (1.7 and 1.9 times, respectively). It appears that the extremely high variation in density among plots masked any effect, and larger sample sizes may have been appropriate. Kolka and others (1998) and Dulohery and others (2000) found green ash survival to be among the highest in their studies and Dulohery and others (2000) found green ash height growth to be greatest.

Two of the three sites (Illinois River and Bankston Fork) had shrub/sapling stage natural regeneration that equaled or surpassed planted tree density. The site with the highest planted tree survival and density (Illinois River) had the greatest density of natural shrub/sapling regeneration as well – five times as great as planted tree density. Bankston Fork showed adequate planted tree survival (32 percent, 316/acre) in spite of dense herbaceous and shrub/sapling cover. Neither of these sites had weed control or release thinning. Dulohery and others (2000) and McLeod and others (2001) found that removing impenetrable (480 to 1,490 stems/acre) willow cover did not improve survival of planted seedlings. Conner and others (2000), however, found that green ash survival was lower in areas cleared of willow. McLeod and others (2000) demonstrated that annual clearing of dense herbaceous cover had no effect on planted bottomland tree survival.

At Rock River, where planted rows were mowed and treated with herbicides annually, trees exhibited an intermediate level of survival (43.5 percent). This could support the theory that removal of herbaceous and woody cover may not improve survival of planted stock. Rock River was the only site that showed a significant decrease in planted tree density between the second and third year. In addition, the adjacent natural regeneration area had a very low density of shrub/saplings and the lowest density of natural tree seedlings (133/ha, 5,750/ha).

The Rock River site experiences winter conditions quite different from the two more southerly sites. Here, mechanical ice breakup in periods of bank full discharge occurred in 2000 and 2001 and are not historically uncommon (Wicker and others 1998, personal observations). At Rock River, random occurrence of ice jams may outweigh other factors in determining survival of planted and natural seedlings alike. Mechanical ice breakup at bank full stage limits development of woodlands in low areas along streams. This phenomenon is capable of killing poles and saplings and is an important cause of mortality for seedlings in floodplains (Scott and others 1997, Smith and Pearce 2000).

SUMMARY

The results of this study indicate that on abandoned agricultural fields in bottomlands in Illinois, planted tree survival may begin to stabilize after 3 years. In this study, planted trees showed adequate to good survival and density (≥ 32 percent, 780 stems/ha) when native bottomland species were selected and planted at a level of 2,220 to 2,470/ha. Green ash appeared to perform best, but differences were not significant. Slower growing, heavy seeded species (pin oak, swamp white oak, and pecan) also performed adequately.

Natural regeneration generally equaled or exceeded density of planted stems, and dense shrub/sapling and herbaceous cover did not appear to detrimentally affect planted stock. Annual mowing and herbicide application did not seem to result in greater planted tree survival, but stochastic events (e.g., ice jam damage) may negate any positive effects.

The site with the most abundant natural regeneration also supported the greatest survival and density of planted trees. However, this may simply demonstrate that the best sites for natural regeneration are also the best sites for planted tree survival. Since dense cover has both positive and negative effects on tree seedlings, the overall effect of natural regeneration on planted stock may be neutral.

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AUGER PLANTING OF OAK SEEDLINGS IN NORTHERN ARKANSAS

Eric Heitzman and Adrian Grell[1]

ABSTRACT.—Planting oak seedlings to regenerate upland oak forests is a promising but untested silvicultural practice in the Ozark Mountains of northern Arkansas. The stony (cherty) soils of the region make it difficult to dig deep planting holes using conventional hand planting tools. In 2001, we planted 1-0 northern red oak and white oak seedlings in 0.5 to 1 acre group selection openings in a northern Arkansas oak stand. Holes were dug with a one-person auger powered by a 4.7 cubic inch chainsaw engine. At the time of planting, average height of red oak and white oak was 3.4 feet and 1.9 feet. After one growing season, red oak and white oak survival was 87 percent and 96 percent. Mean annual height increment for both species was about 9 inches. We recommend using power augers when planting large oak seedlings on stony soils.

Oaks (*Quercus* spp.) are the dominant taxa in the overstory of mature upland hardwood forests in the Ozark Mountains of northern Arkansas. However, silvicultural efforts to regenerate oak naturally are often unsuccessful due to limited establishment and/or recruitment of oak seedlings. In Arkansas, this has lead to the gradual replacement of oaks with non-oak species such as red maple (*Acer rubrum* L.). Such a phenomenon has been reported throughout the midwestern and eastern United States (Abrams 1998).

Various explanations have been offered to explain why oaks are being replaced over such a large spatial scale. Currently, the hypothesis most widely accepted by scientists is that shade tolerant non-oak species accumulate beneath undisturbed mature oak forests and eventually recruit into the overstory following disturbances such as timber harvesting (Johnson 1993, Lorimer 1993, Larsen and Johnson 1998). The accumulation of non-oaks in lower canopy positions has been facilitated by fire suppression activities beginning in the early 20th century (Abrams 1992, Brose and others 2001).

As an alternative to natural regeneration, planting of oak seedlings on uplands may be a viable method of retaining an oak component in future stands. Planting studies have generally been established in clearcuts and small forest openings (e.g., Johnson 1984, Minter and others 1992, Ponder 1995, Zaczek and others 1997, Demchik and Sharpe 1999) or as underplantings beneath a partial overstory (e.g., Johnson 1984, Johnson and others 1986, Gordon and others 1995, Dey and Parker 1997, Bardon and others 1999). These studies indicate that oak seedling survival, diameter growth, and height growth are increased by competition control, the use of tree shelters, and by planting high quality seedlings with well-developed root systems.

Despite these findings, successfully applying research results to an operational level remains a challenge. In northern Arkansas, for example, planting oak on the Ozark-St. Francis National Forest is a new silvicultural practice. In 1999, the Sylamore Ranger District of the Ozark-St. Francis established a 50 acre field trial in which 2-0 northern red oak (*Quercus rubra* L.) seedlings were planted under a shelterwood. Seedling survival after 2 years was estimated to be less than 10 percent (Bob Rhodey personal communication). Forest Service foresters attributed the lack of success to a number of factors, including poor planting technique, low quality nursery stock, and severe summer droughts. Clearly, additional applied knowledge of artificial regeneration on uplands is needed for oak planting to gain widespread acceptance among resource managers.

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Prescriptions have been developed for planting northern red oak and white oak (*Quercus alba* L.) beneath shelterwoods in the Missouri Ozarks (Johnson and others 1986; Weigel and Johnson 1998a, 1998b). In 2001, we established a planting study in northern Arkansas that would begin to develop a protocol for planting oak in small forest openings in the Arkansas Ozarks. The objectives of this paper are to describe our planting methods and to report survival and growth results after the first growing season.

STUDY AREA

Research was conducted in a recently harvested 66 acre mature white oak-black oak (*Quercus velutina* Lam.)-northern red oak stand in northern Arkansas. The stand is on the Sylamore Ranger District of the Ozark-St. Francis National Forest. Soils are Noark and Clarksville very cherty silt loams. These are deep, well-drained to excessively well-drained soils that formed in residuum of cherty limestone (Ward 1983). Slopes range from 5-30 percent. Site index for white oak (base age 50) is 65-70 feet.

In late 1999-early 2000, the stand was commercially harvested using group selection. With this uneven-aged method, approximately one-sixth of the stand will be artificially regenerated with red and white oak seedlings at 10-year intervals in 0.25 to 1 acre scattered openings. We focused our planting activities in the 15 openings created by the harvest. During logging, merchantable trees were felled and saw logs were bucked and removed; tops and other slash were left on site. In January 2001, all stems > 1 inch dbh in the openings (i.e., the unmerchantable trees not felled during harvesting) were girdled by hand and chemically injected with Accord (glyphosate).

MATERIALS AND METHODS

In fall 1999, USDA Forest Service personnel collected red and white oak acorns from 30 phenotypically superior forest trees on the Ozark-St. Francis National Forest. The acorns were sown at the Georgia Forestry Commission's Flint River Nursery in Montezuma, Georgia. Oak seedlings grew at the nursery during the 2000 growing season. The nursery protocol included frequent applications of nitrogen (NH_4NO_3) and is described by Kormanik and others (1994, 2002). The goal of the nursery was to produce tall 1-0 oak seedlings (on average, white oak > 2 feet and red oak > 3 feet) with deep (8 to 10 inches), well-developed root systems.

Large seedlings require deep planting holes. We anticipated the very cherty soils at the planting site would make digging deep holes problematic. Consequently, in December 2000 to February 2001, we tested a number of digging tools at the planting site. We wanted to develop a methodology that would be appropriate both for our research project and for larger, operational planting efforts. Conventional hand tools like planting bars and planting hoes were simply unable to dig an 8-10 inch deep hole in the soil. The same was true of a one-person gasoline-powered auger with a 6 inch diameter auger and a 1.88 $inch^3$ (30.8 cm^3) engine. Planting shovels were somewhat more successful but were extremely time and labor intensive.

Eventually we found our most effective tool consisted of a 6-inch diameter by 31-inch long carbide auger that was connected to a Cannon Tree Planter gearbox and powered by a 4.7 $inch^3$ (76.5 cm^3) chainsaw engine. Smaller chainsaws – including 2.97 $inch^3$ and 3.75 $inch^3$ models – were unable to provide the power needed to drive the auger consistently through rocks and roots. This apparatus, although heavy (about 35 pounds), is well balanced, easy to use, and can be operated by one person.

In late February 2001, the oak seedlings (all 1-0) were graded at the Georgia nursery so as to plant the better quality seedlings. More white oaks were culled than red oaks. The best quality stock of both species was transported to Arkansas and planted in the group selection openings at the planting site on February 24-25 and March 3. Holes were dug with the augers and seedlings were hand planted at 10 foot by 10 foot spacings. Because of the cherty soils, an additional hole was dug several feet from every planting hole to provide soil for packing around the roots of planted trees.

Of the 15 openings that were planted, we randomly selected nine for long-term measurement of oak survival and growth. These openings ranged from 0.5 to 1 acre in size and had northeast aspects. Within each opening, one north-south transect and one east-west transect were established in March after planting was completed. Planted oak seedlings that were on or within several feet of the transects were numbered, tagged, and tallied by species and total height. We measured 35 to 40 seedlings in each opening for a total of 332 seedlings. In October 2001, the tagged seedlings were relocated for 1-year survival counts and height measurement.

RESULTS

About two-thirds of the trees planted were northern red oak (table 1). At the time of planting, the average height of red oak (3.4 feet) exceeded that of white oak (1.91 feet). After one growing season, combined red and white oak survival was 90 percent. Survival averaged 87 percent for the red oak and 96 percent for the white oak seedlings. Red oak remained taller (4.15 feet) than white oak (2.68 feet). On average, both species grew about 9 inches in height during the first growing season. These results are particularly encouraging given that precipitation at the planting site in 2001 was below average (fig. 1).

DISCUSSION

To better assess survival and growth trends, results from most oak planting studies are reported 3 or more years after planting. We recognize the limitations of reporting 1-year results. In fact, the high mortality reported in some long-term studies raises the question of whether planting oaks in clearcuts is a viable regeneration method (McGee and Loftis 1986, Demchik and Sharpe 1999). Nevertheless, we are encouraged by the early survival and height growth of seedlings in this study.

Our results compare favorably with 1-year data reported from oak plantings in clearcuts. In Missouri, for example, 2-0 red oak had 91 percent survival but grew only about 2 inches in height during the first growing season (Crunkilton and others 1989). In Indiana, 1-0 white oak and red oak had 1-year survival of 73 and 80 percent, respectively, and height increments of about 4 inches (Seifert and Fischer 1985).

Table 1.—*One-year survival and height growth of 1-0 northern red oak and white oak seedlings planted on the Ozark-St. Francis National Forest*

SURVIVAL			
Species	Number planted	Number of survivors	Percent survival
Red oak	222	194	87
White oak	110	106	96
Both species	332	300	90
MEAN HEIGHT GROWTH (FEET)			
Species	Height at planting	Height after 1 year	Height growth
Red oak	3.4	4.15	0.75
White oak	1.91	2.68	0.77
Both species	2.88	3.63	0.76

The early seedling success at the study area can be attributed to a combination of factors, including high quality planting stock, low levels of woody competition (trees > 1 inch dbh were killed prior to planting during site preparation), a well trained and closely supervised planting crew, and deep planting holes.

When planting many large seedlings in very stony soils like those in northern Arkansas, we feel power augers are a necessity. Operating the

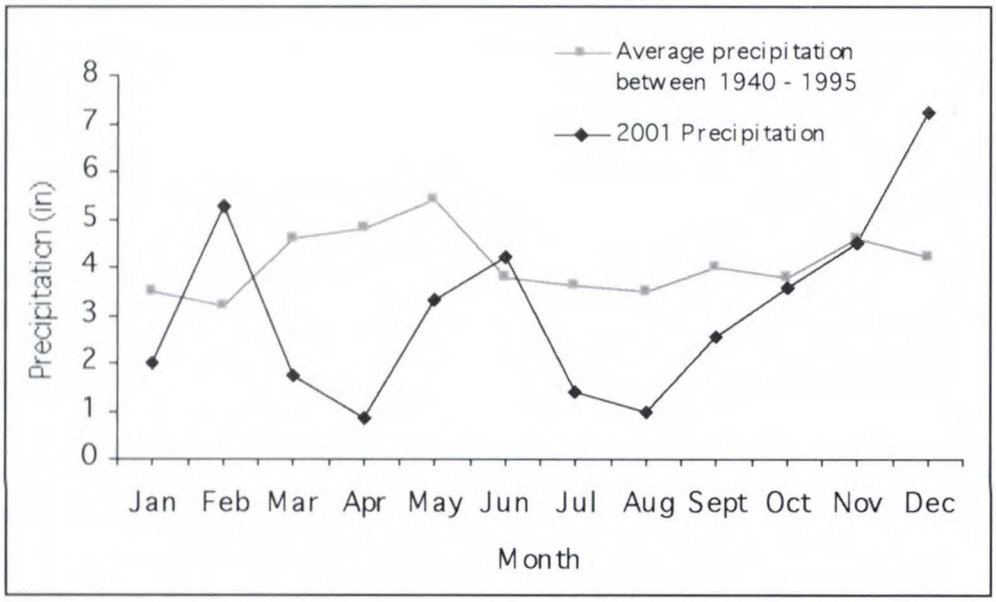

Figure 1.—*Long-term and 2001 precipitation patterns from Arkansas Forestry Commission records for Mountain View, Arkansas, about 10 miles from the planting site.*

type of heavy auger we selected is physically demanding, particularly when, as at the study site, the work requires digging through stones and roots and walking up and down steep slopes through logging slash. Compared to other hand planting tools, augers are relatively expensive. The carbide auger-gearbox-chainsaw equipment we used cost about $1,580. On cherty soils, however, we suggest this is an investment that tree planters need to make to ensure planting success.

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WEED BARRIERS FOR TREE SEEDLING ESTABLISHMENT IN THE CENTRAL GREAT PLAINS

Wayne A. Geyer[1]

ABSTRACT.—Horticultural-type mulches were tested on alluvial sites in two studies to examine survival and growth of black walnut, Scotch pine, and cottonwood seedlings. In one study, black walnut and Scotch pine were established with three weed control treatments using either an annual herbicide or two types of landscape polypropylene fabric barriers. After three years, walnut seedlings had 15 percent greater survival and were 60 percent taller when established with woven landscape fabrics than with Surflan herbicide. Scotch pine seedlings had 25 percent greater mortality with herbicides but no differences in height among treatments. In a second study, cottonwood seedlings were established with a tall fescue sod, a polypropylene weed barrier, black or gray plastic films, cultivation, or herbicide weed control treatment. After 5 years, cottonwood saplings mortality was highest in a grass sod, slightly lower with herbicides, and lowest for the three landscape fabrics and cultivation treatments. Sapling growth was best with cultivation, slightly less for the landscape fabrics and films, and least using herbicides or grass sod for weed control. Mulches appear to be practical for use in tree establishment under environmental conditions of the Great Plains.

Tree seedling establishment typically requires suppression of competitive weeds to achieve high survival and rapid growth. Weed control strategies include cultivation, application of herbicides, ground covers, or weed barrier mulches including plastic or polyethylene sheets and woven landscape fabrics or geotextiles (Appleton and others 1990, Stevenson 1994, Van Sambeek and others 1995). Mulches have several advantages over herbicides and cultivation. Mulches are generally less toxic than herbicides and may not require repeated application. Other benefits can include conserving soil moisture and reducing soil erosion and nutrient leaching (Truax and Gagnon 1993). Although there is scant use of plastic mulches in traditional forestry, landscape mulches are used widely in horticulture. Various new types of plastics and fibers appear frequently on the market and are continually being evaluated (Windell and Haywood 1996).

The early studies with weed barrier materials were done with 1.5 to 6 mil black polyethylene films that were impermeable to water unless mechanically punctured. Woven polypropylene weed barrier fabrics are now being marketed including Sunbelt and Earthmat manufactured by the DeWitt Company in Sikeston, MO. Sunbelt was one of the first permeable weed barrier fabrics marketed. Compared to Earthmat, Sunbelt is slightly thicker (18 and 17 mil, respectively), slightly heavier (110.2 and 101.7 g m^{-2}, respectively), and has a higher infiltration rate (491 and 402 L m^{-2} min^{-1}, respectively). Both are long-lasting and contain carbon black as a UV light stabilizer. Earthmat is about 15 percent lower in price than Sunbelt. The objective of my study was to compare the effectiveness of weed barrier fabrics and plastic mulches to conventional herbicides for controlling competing vegetation in a mixed black walnut and Scotch pine planting and in a cottonwood planting.

MATERIALS AND METHODS

Two studies were conducted in the central Great Plains near Manhattan, Kansas. Precipitation averages about 30 inches per year, with 75 percent coming during the growing season. The planting site was on flat, alluvial, fine sandy loam and silty clay loam.

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In one study, 1-0 (1-year-old) seedlings of black walnut (*Juglans nigra* L.) and Scotch pine (*Pinus sylvestris* L.) were hand planted. Tree seedlings of each species were alternately planted 8 feet apart within ten 300-foot long rows spaced 40 feet apart for a total of 380 trees. The alleyways were planted with various agricultural crops for another study. Each row was divided into three treatment plots 6 feet wide by 100 feet long with 12 or 13 trees per plot.

Weed control treatments were annual application of Surflan herbicide or strips of either Sunbelt or Earthmat woven polypropylene fabric supplied by the DeWitt Company, Sikeston, MO. The area was cultivated before the weed barriers were laid. Seedlings were planted through small X-shaped slits cut in the fabrics. Surflan at commercial label rates was applied each spring over the top of dormant seedlings during the first and second year and along the side in the third year. A randomized complete block design was used with weed barriers and herbicide treatments replicated once in each of the 10 rows or blocks.

In the second study 1-0 seedlings of cottonwood (*Populus deltoides* Bartr. Ex Marsh.) were hand planted 4 feet apart in four 400-foot long rows spaced 12 feet apart for a total of 394 trees. Each row was divided into eight 6 feet wide by 50-foot long plots with 12 trees each. A randomized complete block planting design was used with each row considered a block.

Weed control treatments consisted of plots with solid and punctured 3 mil black plastic mulch, plots with solid and punctured 3 mil gray/black plastic mulch, Sunbelt woven polypropylene fabric, tall fescue sod, annual cultivation, and application of a herbicide using 2 percent Roundup to deaden the sod and post planting application of Gallery (2 lb a.i. per acre) and Surflan (0.75 lb a.i. per acre).

Survival in percent, height in feet, and stem diameter at the ground line in inches of the walnut and pine saplings were recorded at the end of the third growing season and the cottonwood saplings at the end of the fifth growing season. Biomass index is the average oven dried weight for each surviving tree. Plot means were calculated and subject to ANOVA using Duncan's new multiple range test to separate treatment means when significantly different at the $p = 0.05$ level. The two plots of black plastic and the two of gray/black were combined, as there was no statistical difference between the punctured and unpunctuated forms of either plastic film.

RESULTS AND DISCUSSION

In the mixed walnut-pine planting, seedling mortality after 3 years was greater for both species in plots treated with Surflan herbicide than with either Sunbelt or Earthmat weed barrier fabric (table 1). Both species achieved similar survival and heights with either weed barrier material. Unlike pine that had no differences, herbicide decreased walnut seedling height by 40 percent compared to weed barrier fabrics. Although the fabric barriers prevented growth of most weeds, some weeds did grow in the X-shaped slits where trees were planted. Some weeds also grew in soil clods used to hold down the fabric at installation. Limited handwork was done around the trees in the herbicide plots.

Weed barrier fabrics have been shown to improve the soil environment in several similar studies. Polyethylene weed barriers have been shown to increase soil temperatures during the growing season and preserved soil moisture at depths of 6 and 10 inches (Appleton and others 1990, Ham and others 1993, Truax and Gagnon 1993, USDA-Agriculture Research Service 1993). Ham and Kluintenberg (1994) have developed mathematical models relating optical properties and soil contact resistance of plastic mulches to soil heating. The models suggest plastic mulches improve early spring soil temperature. Presumably warmer soil temperatures when soil moisture is still near field capacity leads to improved tree growth.

In the cottonwood study significant differences existed after five growing seasons among weed control treatments for survival, height, stem

Table 1.—*Third-year survival and growth in a walnut/pine planting using three weed control methods*

Species	Treatment	Survival	Height
		— % —	— ft –
Walnut	Surflan	81 b	3.4 b
	Sunbelt	95 a	5.5 a
	Earthmat	92 a	5.3 a
Pine	Surflan	62 b	1.7 a
	Sunbelt	87 a	1.8 a
	Earthmat	83 a	1.7 a

Values followed by same letters within column and species are not significantly different at the $p = 0.05$ level.

diameter, and biomass (table 2). Survival of cottonwood seedlings was very poor in the sod (33 percent) confirming previous observations that weed control is usually required for successful establishment (Boysen and Strobl 1991). Seedling survival was less with the use of herbicides (52 percent) than with weed barrier treatments or cultivation (67 to 89 percent).

Overall, cottonwood saplings had the best growth using cultivation within the tree row to control herbaceous competition (table 2). Although not significantly different, the average height for saplings with the three polyethylene or polypropylene mulches was about 90 percent of the height of the saplings in the cultivated treatment. No differences existed between use of solid and punctured black or gray-on-black plastic film. Sapling height was the shortest using either tall fescue sod or herbicides for weed control.

Cottonwood stem diameter followed a similar pattern to height (table 2); however, diameter for saplings grown with the three polyethylene or polypropylene mulches were 75 to 90 percent of the diameter of the saplings in the cultivated treatment. Using oven-dried weight as a biomass index, cottonwood grown with plastic mulches had about 80 percent of the biomass of the saplings in the cultivated treatment. In contrast, the cottonwood saplings in the tall fescue sod and herbicide weed control treatments had only 50 to 60 percent of the biomass compared to those under the cultivated treatment.

Table 2.—*Fifth-year survival and growth in a cottonwood planting using six weed control methods*

Treatment	Survival	Height	Stem diameter	Biomass index
	— % —	— ft —	— in —	—lbs—
Tall fescue Sod	33 c	15.2 b	2.0 c	6.8 cd
Gallery and Surflan	52 b	15.4 b	2.2 c	5.7 d
Sunbelt Fabric	81 a	21.4 a	3.2 b	15.4 bc
Gray/black Plastic	67 b	22.1 a	3.3 b	14.8 bcd
Black Plastic	89 a	23.7 a	3.8 ab	19.3 b
Annual Cultivation	77 a	24.7 a	4.2 a	29.8 a

Values followed by the same letter are not significantly different at the p = 0.05 level.

Use of herbicides to control herbaceous competition was less effective than expected. As is often the case in field application, the herbicides were applied later in the spring than desired. Late spring application to dry soils in the Great Plains resulted in heavy weed competition and reduced height of both the walnut and cottonwood seedlings. Fabric weed barriers yielded better results than herbicides for several reasons. They can be installed over weeds that may have already germinated and the opaque fabrics will exclude light killing existing vegetation and minimizing germination of weed seeds. With the use of weed barrier fabrics, some weeding may be necessary within the planting hole.

Durability of the plastic mulch in the cottonwood study exceeded life expectancy normally expected under full sunlight because tree leaf litter and grassy vegetation at the barrier edge covered them. Rodents did not appear to damage the tree trunk nor did the plastic girdle the base of the tree.

SUMMARY AND CONCLUSIONS

Weed barrier fabrics have many positive attributes including increasing soil temperatures in the spring, reducing evaporative soil moisture loss during dry periods, and controlling herbaceous competition. Although these UV-resistant fabrics initially may be more expensive to establish, they are practical to use under special conditions such as when planting a limited number of trees, where wind and snow barriers are needed for landscape plantings, or within tree rows that are widely spaced for production of agriculture crops as part of an alley cropping agroforestry practice.

The use of weed barrier fabrics also may be desirable and practical for general forestry. Survival and growth of both walnut and cottonwood were better with weed barrier fabrics than less effective weed control treatments such as herbicides and grass sods. Weed barrier fabrics also may be economically sound, because they require less maintenance in the years following establishment.

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IMPROVING TREE ESTABLISHMENT WITH FORAGE CROPS

Eric J. Holzmueller and Carl W. Mize[1]

Tree establishment in Iowa can be difficult without adequate weed control. Although herbicides are effective at controlling weeds, they may not be desirable in riparian settings and some landowners are opposed to using them. An alternative to herbicides is the use of forage crops to control weeds. A research project was established in 1998 to evaluate the influence of various weed control methods on the growth and survival of a diverse group of tree species.

Seven weed control treatments were used including four forage treatments, one herbicide, one mowing, and a control (no treatment)]. The four forage treatments include hairy vetch (*Vicia villosa* Roth), red clover (*Trifolium pratense* L.), red clover plus red fescue (*Festuca rubra* L.), and red clover plus orchard grass (*Dactylis glomerata* L.). Forage treatments were seeded with cultivated oats (*Avena sativa* L.) as a cover crop.

The tree species included slower growing red oak (*Quercus rubra* L.) and black walnut (*Juglans nigra* L.) from seed and as seedlings and faster growing silver maple (*Acer saccharinum* L.) and two cottonwood (*Populus deltoides* Bartr.) clones Eugenii and Crandon, all as seedlings. In the fall of 1998 and 1999, tree height and basal caliper were recorded for all of the trees. In the fall of 2000 and 2001 height was recorded for all trees, and diameter at breast height was recorded for the Crandon and Eugenii clones.

Survival after the first year on the bottomland site averaged 75 percent for the Crandon clone, 97 percent for the Eugenii clone, 90 percent for silver maple, 93 percent for black walnut, and and 98 percent for red oak seedlings. Survival on the upland site averaged 85 percent for the Crandon clone, 95 percent for the Eugenii clone, 91 percent for silver maple, 93 percent for black walnut, and 94 percent for red oak seedling. Survival was not estimated for the red oak and black walnut from seed. In the spring of 1999, tree seedlings were thinned to approximately 8 to 10 trees per plot to reduce competition.

Trees arising from seed were not thinned to allow for the study of competition in relation to the rate of growth. By the end of the 2001 growing season, the number of Crandon, Eugenii, silver maple, and black walnut seedlings was about equal to the 1999 levels, indicating little mortality. The black walnut from seed had 71 percent survival from 1999 levels on the upland site and 84 percent survival on the bottomland site. Most of the red oak from seed had died with survival from 1999 levels of 16 percent on the upland and 21percent on the bottomland site. The red oak seedlings also faired poorly in the bottomland site with 20 percent survival from 1999 levels, but had 80 percent survival on the upland site.

At the end of the fourth growing season, sapling height on the bottomland site averaged 5.7 m for the Crandon clone, 6.1 m for the Eugenii clone, 2.6 m for the silver maple, 1.5 m for the black walnut from seed, 1.3 m for the black walnut from seedlings, 0.6 m for the red oak from seed, and 1.0 m for the red oak from seedlings. At the end of the fourth growing season, sapling height on the upland site averaged 5.1 m for the Crandon clone, 3.5 m for the Eugenii clone, 1.0 m for the silver maple, 1.2 m for the black walnut from seed, 1.1 m for the black walnut from seedlings, 0.4 m for the red oak from seed, and 0.8 m for the red oak from seedlings.

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After 4 years, the different types of treatment did not appear to influence seedling survival but the type of treatment had a definite impact on height growth with herbicide treatment producing the most growth (2.8 m), followed by the forage crops (2.4 m), the control (2.2 m), and mowing (2.1 m).

Although the herbicide treatment resulted in significantly greater growth than the forage treatments, the 16 percent increase in growth may not be biologically meaningful. In a situation in which tree growth is not a primary consideration the apparent "loss" in productivity due to using forage crops as a way to help establish trees could be a minor concern. It is logical to conclude that given a reasonable level of production from the forage crops, using such crops to help establish trees could be a viable technique.

INCREASING AMOUNTS OF CHEMICAL WEED CONTROL INCREASE GROWTH OF WHITE ASH, WHITE OAK, AND BLACK WALNUT SAPLINGS IN A TALL FESCUE SOD

Jonathan D. Carlisle, Wayne A. Geyer, and J.W. Van Sambeek[1]

Effective weed control in young deciduous plantations is often prescribed to enhance survivability and growth (Bey and others 1975). Chemical weed control often employs broadcast, strip, and spot applications that markedly affect the ground surface area treated. Our study investigates the effect of four levels of treated surface area of a tank mix of glyphosate and simazine and their potential accumulation in the soil on the growth and survival of black walnut (*Juglans nigra* L.), white oak (*Quercus alba* L.), and white ash (*Fraxinus americana* L.) saplings.

The study area is located at the SIU Tree Improvement Center in Carbondale, Illinois. Soil type is a Hosner silt loam soil with a plowpan and a fragipan 18 to 24 inches deep. While not favorable for long-term growth of trees, these soils are appropriate for testing the effects of cultural practices on establishment and early growth of hardwoods. Our study was superimposed on a 4-year-old hardwood planting established in a tall fescue sod. Of the original planting, 28 rows spaced 5 feet apart were selected. Seedlings were spaced 8 feet apart in each row that was planted with three walnuts, then three oaks, and finally three white ash seedlings.

The weed control treatments, listed in increasing amount of controlled area, were no-treatment—0 percent (no-spray), 3-foot diameter circles—17 percent (spot), 24-inch wide strips—40 percent (strip), and broadcast—100 percent (complete). The demonstration area was bi-sected into 2 blocks for treatment replication. Each treatment was assigned to four consecutive rows in block 1 and three rows in block 2. At age 6, 3 walnuts and 19 oak saplings were removed to install a walking trail through the demonstration plot. In addition, 7 ashes, 23 oaks, and 7 walnuts were severely damaged by animals and excluded from the dataset for statistical analysis. A tank mix of glyphosate (2.75 lbs active ingredient per acre) and simazine (4 lbs active ingredient per acre) plus a surfactant was applied under the sapling canopy each spring for 4 years starting at tree age 4, and bi-annually thereafter until the trees were 14 years old. A backpack sprayer with a fan type nozzle applied the tank mix at a rate of approximately 100 gallons per acre.

Seedling (sapling) heights were measured and mortality recorded at tree ages 3 through 7 and 11. During year 5, all saplings were scored for visible herbicide damage on the foliage. Due to the shallow soils and competitive effects following crown closure, height data from year 11 was excluded from the analysis but is shown in the figures to indicate height growth trajectories. Survival is based only on saplings alive at age 3 although mortality in the planting at the time of the first treatment averaged 27 percent for ash, 8 percent for oak, and 11 percent for walnut.

Each species was analyzed separately due to limitations in the experimental design. Height data for each species was analyzed by analysis of variance (ANOVA) as a randomized complete block with covariance. Mean heights for each treatment were adjusted for the variation in sapling heights at the time of the first treatment. Differences between treatments were identified by equality tests of the least squares estimates. This analysis assumes randomness although treatments were not assigned randomly. Survival and herbicide damage data for each species was analyzed utilizing Chi-square tests.

Seedling survival was not statistically different among treatments for any of the three species

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except the spot treatment on the oaks at age 7 ($p = 0.005$). No sapling mortality occurred in the no-spray treatment for white ash, white oak, or black walnut (fig. 1). When averaged over herbicide treatments, sapling survival averaged 96 percent for walnut, 89 percent for white ash, and 79 percent for white oak.

Sapling heights for white oak showed no differences among weed control treatments (fig. 1). Lack of statistical separation of the treatment means is likely due to the small number of trees remaining after those in the walking trail and with animal damage were removed.

Height of black walnut saplings after the second herbicide treatment was greater with complete weed control (4.0 ft, $p = 0.0012$) than the other treatments with no differences among the other treatments. At age 6 and 7, walnut saplings with complete weed control had significantly better growth than with lesser amounts of weed control. After fourth herbicide application at age 7, sapling heights in strip (6.6 ft) were significantly greater than sapling heights in the no-spray (4.6 ft) and spot (4.3 ft) ($p = 0.02$ and 0.04, respectively). The strip and no-spray treatments had a similar effect on height growth.

The percentage of trees with visible herbicide damage (abnormal foliage) in the spray treatments ranged from 17 to 55 percent for white ash, 27 to 66 percent for white oak, and 11 to 33 percent for black walnut but differences among the herbicide treatments (spot, strip, and complete) were not significant.

Saplings with herbicide damage grew more slowly than those without damage in treated areas (fig. 2). At age 5, damaged saplings were 27, 58, and 64 percent shorter than those treated but not damaged for white ash, white oak, and black walnut, respectively. By age 7 damaged trees were still shorter than those treated but not damaged by 30 percent for white ash, 32 percent for white oak, and 51 percent for black walnut. Saplings without weed control were similar in height to saplings that showed herbicide damage at ages 5 to 7 for white ash and white oak. For black walnut herbicide damaged seedlings were shorter than seedlings with no weed control at ages 5 and 6 but not at age 7.

There was little evidence that application of glyphosate and simazine resulted in chronic toxicity leading to increased mortality even with white ash generally considered more sensitive to herbicides than oak or walnut. Except for the spot treatment on the oaks at age 7, there was no increase in mortality between treatments with and without use of herbicides. In contrast, saplings with herbicide damage had slower growth than undamaged saplings through age 7.

In summary, the area covered by herbicides to control tall fescue did affect sapling growth. The complete treatment afforded the best growth for black walnut and white ash saplings. Increased growth in response to increasing treated surface area was first observed after two applications for black walnut and three applications for white ash. Treatment effects for white oaks were inconclusive. When compared to no-spray, the spot treatment did not improve sapling height growth in any of the species. Complete weed control was somewhat more effective at improving tree growth than the strip treatment for both black walnut and white ash. Because of the potential soil erosion that can occur with complete weed control treatment, it cannot be unconditionally recommended.

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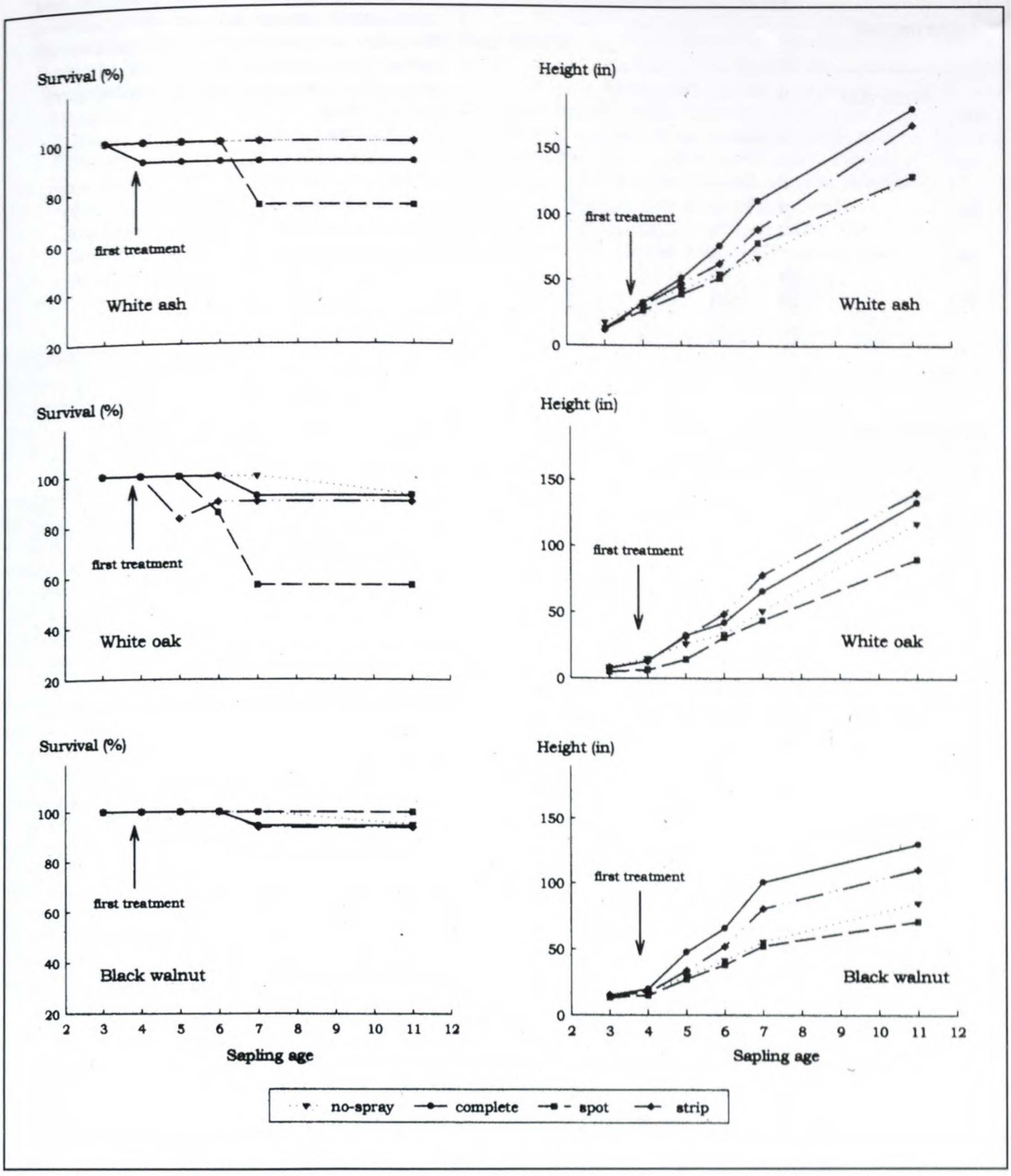

Figure 1.—*Survival percentages and height for white ash, white oak, and black walnut saplings following initiation at age 4 of four weed control treatments that affected percentage of surface area treated.*

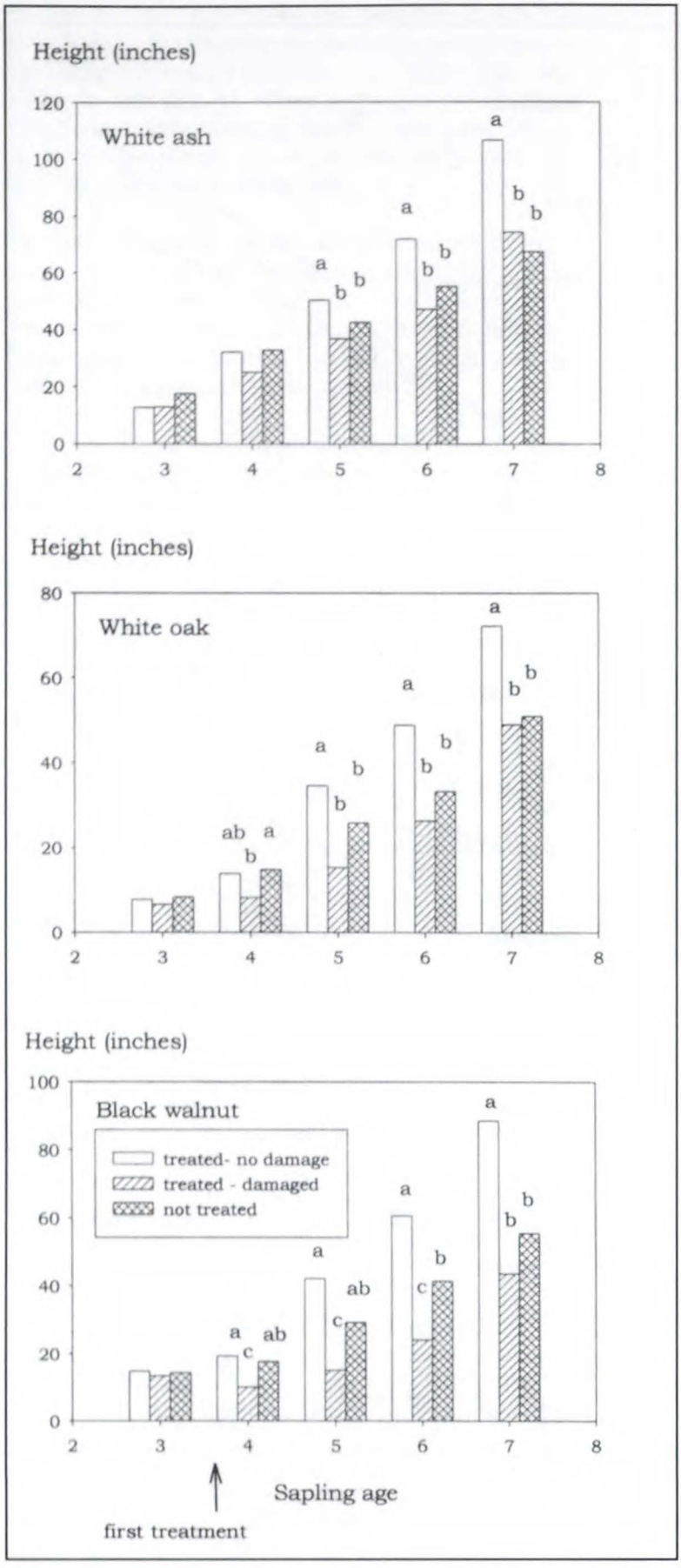

Figure 2.—*Height for white ash, white oak, and black walnut saplings with and without visible herbicide damage at age 5 compared to saplings without chemical weed control. Bars with the same letter are not significantly different at the p = 0.05 level.*

LIRIODENDRON GROWTH ON NATIVE AND STRIPMINE SOILS

W. Clark Ashby[1]

ABSTRACT.—Liriodendron seedlings were planted each spring and fall for 5 years on unmined lands, ungraded spoils, graded to approximate original contour (AOC), and graded to contour with "topsoil" replaced. Survival was greatest on unmined lands and height greatest on ungraded ridge and valley spoil. No Liriodendron seedlings survived on AOC graded mine land. A few small trees survived on graded "topsoil" sites where surface materials had been segregated and replaced. Reclaimed sites with topsoil replaced were highly compacted and had a dense tall fescue grass ground cover. On unmined lands and ungraded spoils, Liriodendron seedlings outgrew black walnut and northern red oak seedlings.

Liriodendron (*Liriodendron tulipifera* L.), also known as tuliptree, yellow-poplar, tulip-poplar, white poplar, or whitewood, is a valuable, fast-growing eastern hardwood that produces large quantities of seed from which planting stock is readily available. From its natural distribution related to climate, parent material, relief and biota and from past planting studies, Liriodendron is known to be more site sensitive than many of its associates. This paper focuses on reinforced or new understanding of Liriodendron's site requirements to guide future plantings, and on its potential for evaluating planting sites in stripmine reclamation.

Surface-mined lands are of importance because of their large and expanding extent, non-vegetated post-mining condition, and potential for highly productive tree growth when site conditions and choices of species for planting are fitting. Different kinds of post-mining landscape and soils are found depending on applicable government regulations when mined. Greater understanding of plant/rooting medium relationships is needed for successful management of new lands after mining. A fuller description of the mining process and resultant site conditions in southern Illinois is given in Ashby (1999).

My objectives were to evaluate

1) how well does Liriodendron grow on mined land,
2) what planting practices are useful for reforesting mined lands with Liriodendron,
3) how does Liriodendron seeding survival and growth on three kinds of mined lands compare to seedlings on unmined reference areas, and
4) how does Liriodendron seedling growth compare to other tree species used in reclamation?

GROWTH ON MINED LANDS

Related to seed source, Liriodendron has readily invaded mined land in eastern Kentucky and Tennessee (Rafaill 1988). Liriodendron saplings dominated on one unreclaimed mountainside contour mine (833 stems/ha). On two mines reclaimed by planting black locust (*Robinia pseudoacacia* L.) plus herbaceous legumes and grass, the dominant invading species were red maple (*Acer rubrum* L.) and Liriodendron with 300 and 167 stems/ha, respectively.

Limstrom (1960) summarized numerous species trials on mined lands in the midwest from a series of USDA Forest Service research plots.

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Liriodendron was successful on deep well-drained spoil in the pH range 4.5 to 7.5, with best growth at near-neutral pHs (fig. 1). The N, P, K content of leaves from 4-year-old Liriodendron on both not graded and graded areas of three strip-mine areas in Ohio compared favorably with that of leaves from trees growing on unmined land (Finn 1952). Stony soils on stripmines have been productive for tree growth and crops (Ashby and others 1984).

Although Liriodendron is commonly dominant on better soils in ravines of the southern Illinois hill country, it is conspicuously absent from the fragipan and other soils of adjacent glaciated counties surface mined for coal. Planted Liriodendron is one of the fastest-growing species on ungraded minesoils in these counties and reached small sawtimber size in 50 years. Heights and diameters at breast height (DBH) were measured more than one time for three of eight sets of plots (table 1).

Liriodendron has failed or grown poorly on other USDA Forest Service plantings, especially those using other reclamation practices. These plantings failed because of poor drainage, tight clay soils, continuous rock surfaces, dense overhead or ground cover, and other causes atypical of good sites for Liriodendron in its natural range (Limstrom 1960).

Table 1.—*Height/DBH of Liriodendron at several ages related to topography and pH at planting on stripmines. Boyce and Neebe 1959, Ashby 1996, Unpublished data.*[1]

	AGE				
PLOT	10	22	30	40	47
			m/cm		
Unleveled					
pH 3.4	3/-	-/-	-/17	-/-	28/33
pH 5.7	-/-	18/21	-/-	30/30	-/-
Leveled					
pH 6.1	3/-	-/-	-/28	-/-	27/36

[1]The pH 5.7 planting was under decadent black locust.

PLANTING PRACTICES

Varied planting practices to improve reclamation success have been used in conventional forestry and in reclamation. We evaluated their usefulness to cope with new and difficult problems after passage of a 1977 federal reclamation law (P.L. 95-87, 1977). Our research plan included successive spring and fall plantings from 1978 to 1982 on three types of reclaimed lands and on unmined lands. These studies were carried out on the Sahara Coal Co., Inc. Mine No. 6 in southeastern Illinois near Harrisburg.

Figure 1.—*Cross-section of stripmine spoil banks at pH 5.1 in southern Illinois with 36-year-old shortleaf pine (*Pinus echinata *Mill.) on the sides and white oak (*Quercus alba *L.) on the left, and 23-year-old Liriodendron in the center. The tree by the person is 22 m tall, 30 cm DBH. Mean Liriodendron stand height was 18 m and DBH was 21 cm.*

Planting methods were similar to those of earlier USDA Forest Service plantings. Replicated plots were planted to 50 trees for each of 20 species at 2.5 m spacing of trees within and between rows. Liriodendron was included as a known site-sensitive tree to characterize limiting conditions and hopefully to find a way to use it in reclamation plantings. Black walnut (*Juglans nigra* L.) and northern red oak (*Quercus rubra* L.) were included for comparison with Liriodendron. Species were randomly assigned to tree rows in a plot so that the nearest neighbors to Liriodendron varied.

Liriodendron seedlings (1-0) from the Union County State Tree Nursery were kept in cold storage until planted. All seedlings were measured for seedling diameter, stem and root length, and graded for quality and dormancy to select for seedling uniformity. Only one-half to two-thirds of each lot of seedlings were used for these plantings. Roots were trimmed to 20 cm long and laterally pruned if needed for planting with a planting bar.

Seedling lots differed in seedling size and dormancy from planting season to planting season. Differences in average stem caliper varied from 11 mm in fall 1981 to 6 mm in spring 1981. Average stem caliper was not consistently related to survival. Root lengths were atypically short in fall 1978, spring 1979, and spring 1980 and were only 15 cm long in spring 1982.

Spring-planted seedlings had been held in cold storage for approximately 6 months. Fall-planted seedlings had been held in cold storage for a few weeks to months. Survival and growth of fall-planted dormant stock were similar to that for seedlings planted the following April. Newly-lifted Liriodendron seedlings planted in September 1978 and September 1979 had no or low survival in contrast to good survival for stored seedlings planted in November 1980 and early December 1981. Frost heaving was not reported.

Seedling counts were taken in years 1 and 3 after planting. Surviving trees were measured for height in 1988 and for height and DBH in 1993. Ages in 1993 thus varied from 16 to 11 years for the successively-planted plots. From 1979 to 1982 the seedlings planted the previous fall and the spring-planted seedlings had simultaneous growing seasons. Survival was equal to or greater for spring- than fall-planted seedlings except for the excessively wet spring in 1981 when the planting was also in late May rather than April (fig. 2).

Starting with the 1981 growing season, all vegetation in a 1.5 m circle around tree seedlings planted the previous fall and that spring was sprayed with a contact and pre-emergence herbicide mixture of glyphosate (Roundup®) and simazine (Princep®). Tree seedlings were shielded during spraying. Herbaceous weed control lasted 2 to 4 years. Other than exceptionally vigorous seedlings planted in spring 1979, survival of Liriodendron seedlings planted in 1981 and 1982 on unmined land and on ungraded spoil with chemical weed control had overall greater survival than seedlings planted in 1978, 1979, and 1980 without chemical weed control (fig. 2). Sapling heights on the plots with chemical weed control, especially on the ungraded plots, were greater than expected for the youngest trees (fig. 3).

ROOTING MEDIA EFFECTS

The plots for the present study were on unmined lands, ungraded pre-law ridge and valley spoil, spoil graded to approximate original contour (AOC), and "topsoil" or AOC graded spoil with replaced surface materials as required under P.L. 95-87. The replaced surface materials of pre-mining eroded old fields were chiefly unproductive subsoils. These sites were only cosmetically similar to the adjacent unmined lands in the present study. Regulatory authorities coined the terms "Topsoil" and "Prime Farmland Fragipan Soil" for the landscapes graded with replaced surface materials. Agronomic practices of fertilization and planted legume and grass ground cover had been carried out as required on both the graded and "topsoil" sites. Dense stands of tall fescue (*Festuca arundinacea* Schreb.) had become dominant on most planting sites.

Survival

Liriodendron survival varied greatly depending on the year of planting and on the rooting media (fig. 2). With few exceptions most mortality took place in the first 3 years after planting. No plantings of 1,000 seedlings each on graded AOC plots survived for any planting season from spring 1978 to fall 1982. Overall survival on the ungraded plots averaged 28 percent (280 of 1,000 trees planted) and 52 percent (520 of 1,000 trees planted) on the unmined plots. Only seven small trees of the 500 planted on the "topsoil" plots from fall 1980 to fall 1982 survived to 1993 out of the 34 that had survived to age 3.

Features of the two graded spoils that likely resulted in little or no survival are the combined

effects of compaction and dense ground cover, both of which are otherwise recognized as limiting factors for Liriodendron (Burns and Honkala 1990). Bulk densities at several plot locations ranged from 1.70 to 1.94 with an average 1.80 g/cm^3 compared to 1.27 g/cm^3 for the ungraded spoil. A compaction level of 1.60 g/cm^3 is commonly considered limiting for root growth (Josiah 1986). Biological factors affecting Liriodendron likely included competition especially for water and known allelopathic toxicity from the dense tall fescue cover (Peters and Zam 1981, Rink and Van Sambeek 1985).

Survival on the unmined plots in the present study has been greater than on any of the mine soils related to undisturbed surface soil properties and organisms and partial tree cover at time of planting. The unusually wet 1981 spring season (planting number 7 in figure 2) was an exception with poor survival. Those plots were on a bottomland soil and excessive moisture was likely the cause.

Limited survival during some planting seasons on the ungraded ridge and valley banks may have been caused by drought. The two driest years, 1978 and 1980, had the greatest mortality. Root growth of seedlings planted those years (plantings numbers 1, 4, and 5 of figure 2) on the relatively coarse-textured minesoil may have been too slow to reach deeper moisture reserves. The few surviving small Liriodendron trees on the compacted "topsoil" plots were on a lower-lying area of the mine.

No pre-mining soil features except textures were evident on the "topsoil". The rooting medium was commonly unproductive B-horizon fragipan with occasional gray layer materials. Both the replaced surface materials and the underlying graded minesoil sites were highly compacted. The tall fescue cover with interspersed old-field broomsedge (*Andropogon virginicus* L.) and goldenrod (*Solidago* L. spp.) was less dense on the "topsoil" plots than on the graded AOC plots.

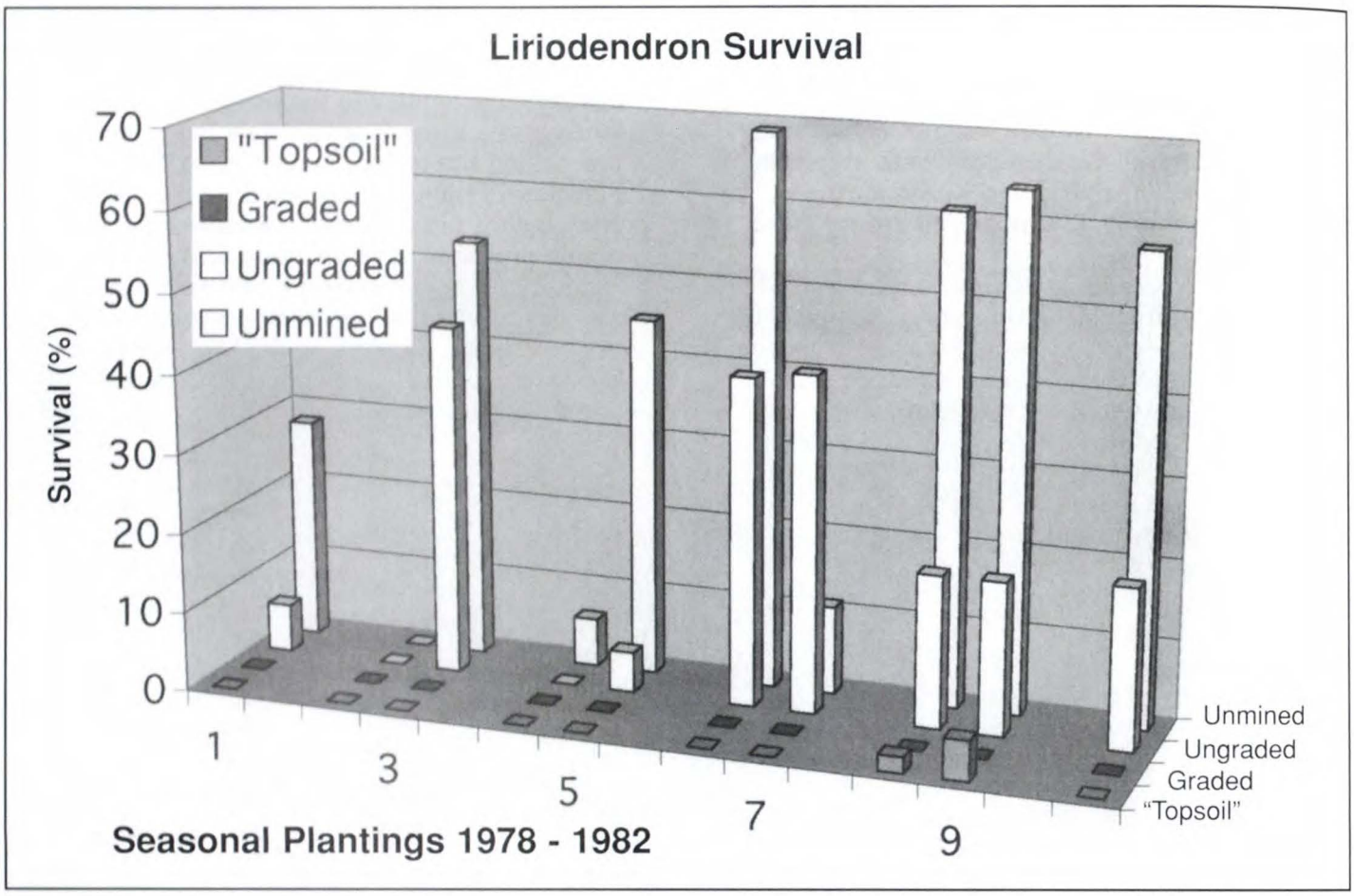

Figure 2.—*Liriodendron survival in 1993 related to rooting media and initial growing season on graded spoils with "topsoil" replaced, spoils graded to original contour, ungraded ridge and valley spoils, and adjacent unmined land. For each paired column the odd-numbered (right) one was planted that spring and the even-numbered (left) one was planted the previous fall. Column one was spring-planted in 1978 and column 10 fall-planted in 1982.*

Height

Liriodendron height related to planting season showed less variation than did survival (fig. 3). Average tree heights were typically greatest on the ungraded ridge and valley spoils, variable on the unmined lands, and very low on the "topsoil" sites. The mandated poor quality and compacted rooting medium together with the tall fescue and other cover on the "topsoil", even with chemical weed control, resulted in an average height growth of only 1.7 m after 12 years (plantings numbers 8 and 9 of figure 3). Required fertilization, commonly not recommended for forest tree plantings, likely accentuated the deleterious effects of the ground cover.

Trees tended to be taller on the ungraded spoil than on the unmined land that was chiefly Hosmer silt loam, Typic Fragiudalfs Alfisols (Miles and Weiss 1978). Plantings numbers 5 and 6 were on Hoyleton silt loam, Aquollic Hapludalfs Alfisols, and the number 7 was on Banlic silt loam, Aeric Haplaquepts Inceptisols. Except for the planting on the Banlic silt loam, which has the highest estimated annual growth of timber for these soils (Fehrenbacher and others 1978), the ungraded minesoils had Liriodendron heights equal to or greater than the unmined reference areas. After 11 years on our ungraded mine spoil (number 10 in figure 3), tree heights averaged 5.1 m compared to only 3.1 m after 10 years in other studies on productive sites (table 1). Although tree age ranged from 11 to 16 years, height growth for fall-planted seedlings was similar possibly because only the young plantings received chemical weed control.

Liriodendron outgrew black walnut in eight comparisons of means on the ungraded plots, and in four comparisons on the unmined plots where black walnut survived (table 2). Liriodendron heights were relatively similar to those of northern red oak on the ungraded spoil and were greater on the unmined lands. Red oak on "topsoil" was about twice the height of the Liriodendron and black walnut trees that survived. Bottomland oaks are now recommended from several studies for planting on "topsoil" sites. The present study is an opportunity to

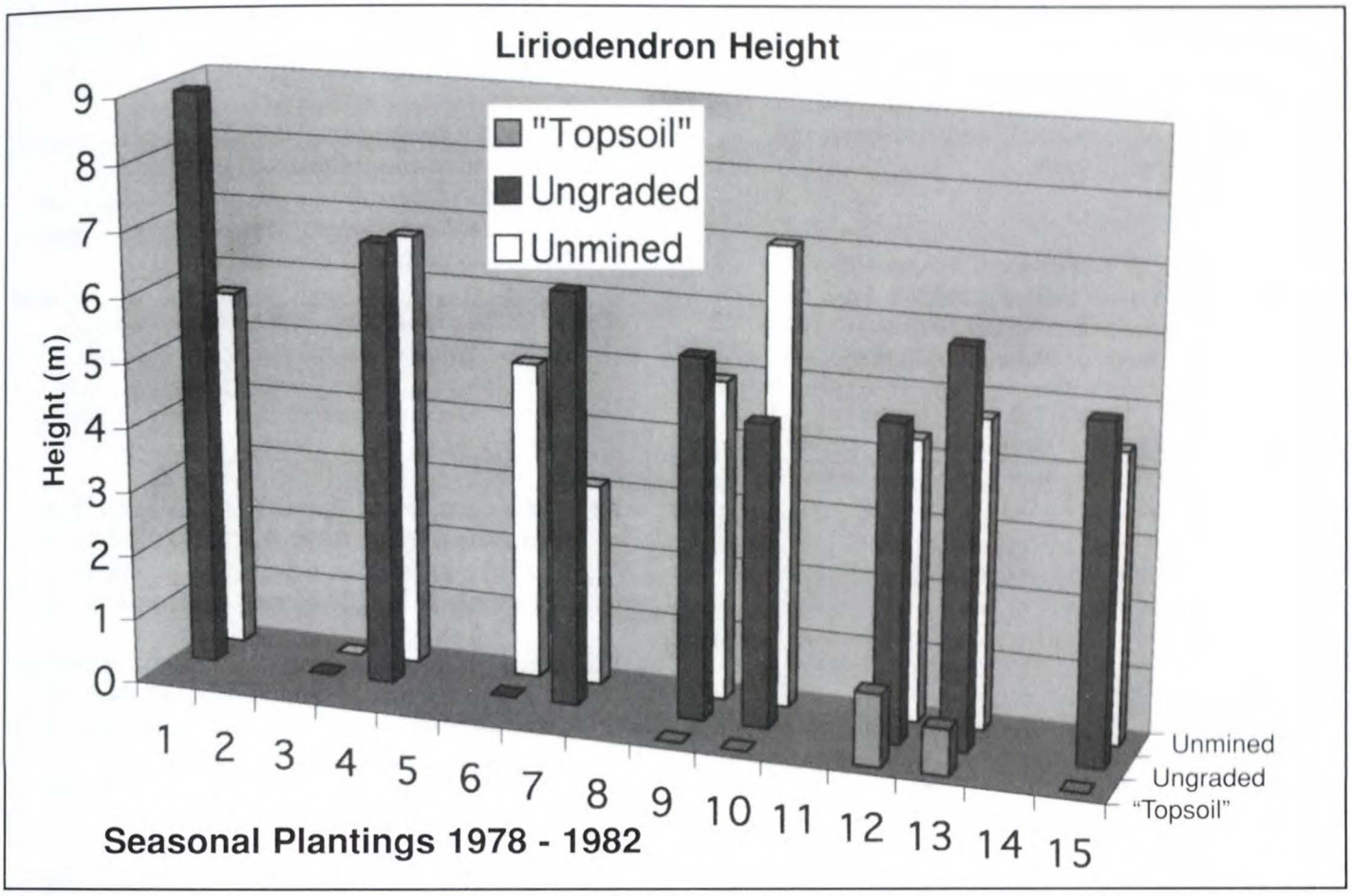

Figure 3.—*Liriodendron height related to rooting media and initial growing season. The mine spoil and planting designations are the same as in figure 2. No Liriodendron survived on the spoil graded to original contour plots.*

test the recently published equations to predict forest productivity for Liriodendron, black walnut, and northern red oak in southern Illinois (Woolery and others 2002).

CONCLUSIONS

1) Survival was lower and average height greater for planted Liriodendron on ungraded stripmine spoil than on adjacent unmined lands where it did not occur naturally.
2) Poor seedling survival on graded AOC or "topsoil" sites was associated with excessive compaction and dense tall fescue cover.
3) Seedlings planted in September failed, probably because they were not dormant.
4) Survival and growth of seedlings were similar when planted in early December or in April.
5) Percent survival was a poor predictor of height growth.
6) Better quality control in nursery production of Liriodendron is recommended, although effects of seedling size were not marked.
7) Benefits for chemical weed control on Liriodendron survival or height growth were confounded by age differences.
8) Only the ungraded spoil matched the productivity levels of Liriodendron on the unmined reference fields.
9) Liriodendron was a relatively better choice for ungraded and unmined lands, black walnut for AOC graded, and northern red oak for "topsoil" sites.

Table 2.—*Height differences in 1993 between Liriodendron and black walnut or northern red oak planted each season from spring 1978 to fall 1982 on ungraded minesoil or unmined rooting medium*

Planting season	Ungraded		Unmined	
	Lir/BW	Lir/RO	Lir/BW	Lir/RO
	m			
Spr. '78	3.6	0.1	2.1	–
Fall '78	–	–	–	–
Spr. '79	2.6	–	–	2.2
Fall '79	–	–	–	4.0
Spr. '80	2.4	–	-0.3	0.1
Fall '80	1.0	–	-0.3	–
Spr. '81	0.9	6.1	-0.4	5.2
Fall '81	1.1	3.4	1.0	2.1
Spr. '82	1.9	2.2	0.8	2.5
Fall '82	1.5	2.4	1.0	3.1

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THE HARDWOOD TREE IMPROVEMENT AND REGENERATION CENTER: ITS STRATEGIC PLANS FOR SUSTAINING THE HARDWOOD RESOURCE

Charles H. Michler, Michael J. Bosela, Paula M. Pijut, and Keith E. Woeste[1]

ABSTRACT.—A regional center for hardwood tree improvement, genomics, and regeneration research, development and technology transfer will focus on black walnut, black cherry, northern red oak and, in the future, on other fine hardwoods as the effort is expanded. The Hardwood Tree Improvement and Regeneration Center (HTIRC) will use molecular genetics and genomics along with traditional breeding, selection, and vegetative propagation to produce improved hardwoods, deployment systems and management strategies for these fine hardwood species. In addition, new knowledge of population genetic structure that will guide regeneration efforts through increased productivity and maintenance of genetic diversity in Central Hardwood forests.

The mission of the Hardwood Tree Improvement and Regeneration Center (herein after called "HTIRC"), is to advance the science of hardwood tree improvement and genomics in the Central Hardwood region of the United States by:

1) Developing and disseminating knowledge on improving the genetic quality of hardwood tree species;
2) Conserving fine hardwood germplasm;
3) Developing elite hardwood trees for restoration and regeneration of sustainable hardwood forests and riparian zones for production of forest products and maintenance of genetically diverse ecosystems; and
4) Developing recognized and respected science leaders in forest genetics.

ORGANIZATION DESCRIPTION AND CHARACTERIZATION

The HTIRC program is a regional collaborative research, development, and technology transfer effort. This partnership includes the USDA Forest Service North Central Research Station, National Tree Seed Laboratory, and Northeastern Area State and Private Forestry; Purdue University Department of Forestry and Natural Resources; Indiana Department of Natural Resources Division of Forestry; Indiana Hardwood Lumbermen's Association; National Hardwood Lumber Association; Indiana Forestry and Woodland Owners Association; American Chestnut Foundation-Indiana Chapter; Walnut Council; and the Fred M. van Eck Forest Foundation. It is unique in several aspects:

1) The HTIRC has a regional focus on states comprising the Central Hardwood region;
2) It is a true partnership of federal, state, university, industry, and landowner groups who contribute financial support and advice; and
3) It will focus on the development of basic knowledge and technologies in hardwood tree genomics, improvement, and regeneration for tree nurseries, industry, agencies, and landowners.

HTIRC is located at Purdue University because of its role in the Midwest as a recognized center for agricultural genomics research. It is housed in the Whistler Hall of Agricultural Research with biotechnology and genomics faculty from six departments in the School of Agriculture. This co-location of genetics faculty is intended to stimulate the cross-fertilization of research ideas and multidisciplinary research. In addition, because of the high standards of achievement and international recognition of its faculty, high quality undergraduate and graduate students should be drawn to the various programs represented in Whistler Hall.

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OPERATING ENVIRONMENT

The idea for HTIRC was conceived in 1998 because of a perceived void in hardwood tree improvement research in the Central Hardwood region of the Midwest and Northeast. The birth of HTIRC occurred at the same time that the region was experiencing a severe shortage of hardwood seedlings estimated anywhere from 25 to 50 million trees annually and that shortage was increasing 20 percent annually (unpublished survey of Midwest state foresters). In addition, the majority of seedlings being produced in state nurseries were of unimproved genetic source because nurseries rely upon seed collectors to collect and transport seed to the point of purchase at the nursery. Thus, the majority of seedlings being produced had unknown fitness for sustainable forestry and unknown genetic diversity.

The hardwood industry was also concerned about the future quantity and quality of the resource for its lumber and manufacturing sectors. Due to political and social pressures, federal forests have significantly reduced the volume of hardwood timber that is being harvested annually. Small private woodlots that supply the majority of hardwood timber in the region are not being managed in a sustainable manner, ownership is not continuous over numerous rotations to insure sound forest management, and many woodlots are being converted to residential and recreational uses. In addition, the diameter of timber harvested today continues to be smaller than what it has been due to shorter rotations. Last, the hardwood industry was concerned that it was not taking advantage of new biotechnologies that could increase wood production through tree improvement activities that improve wood quality, growth, and pest resistance.

The professional forest community was also concerned about loss of genetic quality in remaining hardwood woodlots and natural forests. They felt that trees currently being managed for future timber harvest do not have the same desirable traits for straightness and vigor and that past forest harvest practices of continually taking the "best" trees may have resulted in loss of genetic quality of the remaining germplasm.

INTERNATIONAL ECONOMIC, SOCIAL, AND POLITICAL ENVIRONMENT

Human population growth continues to expand rapidly, and consumer demand for quality hardwoods, at some point, will outstrip the region's ability to produce it unless consumers are willing to accept substitute materials. Much of the US and European demand for hardwood lumber is currently met within the United States' Northern and Central Hardwood zones, and hardwood production has not shifted to developing countries in any significant manner. However, the time may come when this pattern of production for world markets can no longer be maintained because of decreased supply of quality hardwoods in North America and the environmental consequences of heavy timber extraction on natural environments.

In the Central Hardwood region, water quality has been degraded by agricultural intrusion along waterways, and flooding has further degraded these riparian zones along major rivers and their tributaries. A significant effort is being made to restore hardwood trees and other native vegetation in these riparian zones. For the most part, unimproved trees are being used, and the knowledge of how to restore these degraded areas has been lacking, resulting in failure of many plantings. The opportunity exists for this significant portion of the land base to be a future site for quality hardwood regeneration although political forces may influence that reality.

Funding for many conservation plantings comes from the Conservation Reserve Program (CRP), Conservation Reserve Enhancement Program (CREP), and Wetland Reserve Program (WRP). These programs currently account for the majority of hardwood tree planting and they are increasingly focused on improvement of water quality and wildlife habitat. The genetic characteristics of the trees being planted are unknown and the potential exists for genetic failures as these plantings mature.

Many consumers do not differentiate between a plantation forest and a natural forest. They readily accept that agronomic crops are grown for the purpose of food production rather than for regeneration of annual vegetation. Conversely, consumers have certain aesthetic and spiritual values for trees whether they are in an urban, plantation, or natural forest environment. If these values do not change, then plantations may not be viewed as an acceptable alternative method for growing highly productive crop trees which would then allow the country to maintain natural forests as preserves for biodiversity and recreation. These attitudes exist despite the fact that only a marginal set-aside of land currently in forest land cover

would produce all of the wood necessary to meet human demands for wood consumption (Sedjo 1999).

Few cities have resolved the conflict that occurs when urban sprawl imposes upon adjacent rural and forest environments. As community planners continue to struggle with these issues, more forest and farmland continues to be converted into residential use. The increase in economic affluence of American households will continue to put pressure on the urbanization of these environments because the desirable qualities of forest environments for residential communing.

RELATIONSHIP WITH OTHER ORGANIZATIONS

For quality veneer, central and northern hardwoods are more desirable than southern hardwoods because of slow, more even growth. In addition, the supply of European hardwoods is limited. Because of this, the HTIRC will hold unique importance within the central United States to develop superior hardwood trees for markets that supply developed nations. There are few other hardwood tree improvement research centers in the world and few individual hardwood research programs that have a mandate to satisfy a regional clientele and are financially supported to undertake a long-term program in tree improvement.

HTIRC will be unique because of Purdue University's relationship with the Donald Danforth Plant Science Center in St. Louis, Missouri. The Danforth Center is a partnership of the Missouri Botanical Garden, Monsanto, Purdue University, University of Missouri-Columbia, University of Illinois, and Washington University in St. Louis. Its mission is to increase understanding of basic plant biology and apply new knowledge to sustain productivity in agriculture and forestry. HTIRC should benefit from its access to world-class plant scientists, and its ability to collaborate in the graduate education of its students.

HTIRC will be vertically integrated with molecular and classical geneticists, tree physiologists, silviculturalists, and nursery and regeneration specialists. Its strength will be its ability to perform basic, applied, and developmental research so the basic genetic knowledge that is created will be delivered to industry and private landowners in value-added products rather than knowledge that only benefits the scientific community.

HTIRC will not be the single or sole institution performing hardwood genetic research. The region has many outstanding scientists who perform valuable basic and applied research on various species, and it will be essential that the whole hardwood scientific community remains viable to meet these research needs. In addition, HTIRC does not plan to employ pathologists, entomologists, biochemists, enzymologists, economists, and the other scientific disciplines that are necessary for research collaborations to provide scientific data for evaluation of the ecological and environmental fitness of HTIRC products. Productive working relationships with scientists from other institutions are necessary for the success of the Center and for maintaining healthy, sustainable forestry in the Central Hardwood region.

FUNDING

Annual funding appropriations are expected from the USDA Forest Service North Central Research Station, Northeastern Area State and Private Forestry, Purdue University Department of Forestry and Natural Resources, the Fred M. van Eck Forest Foundation, the wood products industry and various landowner associations.

As products are developed and discoveries are made, protection of intellectual property and products may be patented or trademarked. Partners and HTIRC endowment would benefit from these sources of funds.

STRATEGIC DIRECTIONS

HTIRC has six (6) strategic directions:

1) Improve the genetic quality and regeneration of fine hardwoods, including black walnut, black cherry, and northern red oak, through application of classical breeding, genomics, molecular markers, genetic modification, advanced vegetative propagation, seed production technologies, and silviculture.
2) Establish a highly credible hardwood genetics research center that will be recognized as a leader in forest genetics, and thereby become a leading graduate education and training facility for future scientific leaders in hardwood forest genetics.
3) Hire and nurture pre-eminent scientists who will build the credibility of the research program and be highly competitive for federal research grants.
4) Establish the Martell Research, Education, and Conference Center as a significant site for education and training of consulting foresters, nursery practitioners, and landowners in nursery management and hardwood culture.

5) Communicate, convey, and market the work of HTIRC in order to be perceived as the pre-eminent international center for hardwood genomics and biotechnology.
6) Secure funding for an endowment to insure long-term organizational stability, provide for operating support of the research program, and establish funded research positions within the Center.

OBJECTIVES

The following are objectives for implementing the strategic directions:

1) To develop research and technology transfer programs that provide knowledge for management and maintenance of sustainable, genetically diverse natural forests and highly productive domesticated trees for plantation hardwood crops that provide a wide array of products.
   - For black walnut, black cherry, and northern red oak: develop genetic maps, molecular markers, tissue culture and genetic engineering technologies, advanced seed orchard, and seed handling technologies, breeding orchards, and experimental nurseries for production of elite families and cultivars, identification of superior seed trees, and assessment of genetic quality and diversity in natural stands.
   - Assess the need and develop a regional hardwood breeding and seed orchard cooperative for Central Hardwood seed zones as desired by state tree improvement cooperators.
2) To take leadership in documentation of hardwood research discoveries and dispersal of knowledge by hosting scientific conferences, symposia, workshops and field days, and publish books, proceedings, and brochures that convey this knowledge to a wide array of end users.
3) To provide for annual evaluation and other periodic review of HTIRC programs to ensure that the mission and vision remain focused and relevant.

ACCOMPLISHMENTS AND FUTURE PLANS

Since establishment in 1998, staff at the HTIRC have developed tissue culture systems for northern red oak and black walnut; developed over 500 black walnut and northern red oak molecular markers for use in genetic mapping, assessment of genetic diversity and marker-assisted breeding; established new breeding and seed orchards for black cherry and black walnut; established a field repository for fine hardwoods with unique genetic traits; initiated studies to develop trellis seed orchards; established a block of canker-resistant butternut; and established two field trials for screening Midwest-adapted blight resistant American chestnut. In the remainder of this paper, we will focus on current work in black walnut genetics and vegetative propagation.

Black Walnut Genetics

Research in black walnut genetics at HTIRC has been aided immeasurably by the publications, plantings, and meta-data generated by previous walnut researchers in the USDA Forest Service and at Purdue University. An outstanding example is the longitudinal data collected over the last 20 years on 80 black walnut families planted at the Southeast Purdue Agricultural Center (SEPAC, Jennings County, IN) as 1-0 seedlings. These seedlings were derived from open-pollinated seedlots of superior selections identified by the USDA Forest Service as part of study FS-NC-1151, 82-03. The trees planted at SEPAC were half-sibs of trees planted at the Kellogg Forest near Belmont, Michigan, as part of a cooperative agreement with Michigan State University, and at Big Creek, Hardin County, in southern Illinois. These resources permit us to test genetic theories on mature stands with known genetic and management histories.

Several geneticists have theorized that selection among walnut progenies at a relatively young age (4 to 6 years) could be effective at improving rotation height and diameter (Kung 1975, McKeand and others 1979, Rink 1984). This conclusion was based on relatively small data sets, quantitative genetic theory, results from other species, and data from young plantations (≤ 10 years). Two of these authors later suggested that selection any time after age 8 would be acceptable (Rink and Kung 1995) based on a 20-year-old progeny test. Evaluation of the data taken at SEPAC (FS-NC-1151) amplifies this finding. Results from SEPAC reveal a considerable amount of rank shifting for height and diameter at the level of individual trees, as well as among and within families, before age 10 (Woeste, unpublished data).

While results of studies on early selection in other species are important methodologically, the application of selection theory to walnut requires sensitivity to the biological and reproductive characteristics of the species. Walnut propagules of known ancestry have a high value because they are rare and/or expensive to produce as a result of low seed yield (compared to

many species), the unpredictability of seed bearing, poor seed viability after one year of storage, long tree juvenility, the difficulty of producing controlled crosses, and the high cost of grafting.

It may be more cost effective to delay selection until rank shifting within and among walnut families has been reduced to very low levels. The previously mentioned high value of propagules means that the marginal cost of an additional year or two of evaluation may be much smaller than the cost of prematurely discarding a valuable family. Lambeth (1980) cites similar reasons for delaying selection in the Pinaceae, and notes that, in general, the most efficient time for selection increases as the rotation age increases. Lambeth suggested a selection age of 6 to 8 years for the Pinaceae, which have a rotation age of 30 to 40 years, whereas the rotation age for walnut is often twice these values.

Large increases in heritability between ages 10 and 15 observed in the SEPAC planting indicate that selection after age 10 might permit breeders to maximize the value of observed differences among individual trees, although these increases in heritability have not been seen at all study sites. Rink (1984) found the heritability of height growth increased steadily after age 4 until age 10. Rink and Kung (1995) reported similar results, with heritability holding steady or decreasing slightly after age 10. Citing Namkoong and Conkle (1976), Rink predicted that heritability would continue to increase until a plantation reached a mature phase of growth, which he equated with crown closure. Perhaps that was why Rink and Kung found that heritability in their study site at Pleasant Valley leveled off at age 10. Crown closure at Pleasant Valley was as early as age 5. Trees at Pleasant Valley increased their diameter at a moderate 0.27 inches per year, and Rink and Kung reported a rising variance for block throughout the life of the plantation due to competition among seedlings.

The SEPAC planting was maintained with a minimum of crown closure for the entire 20 years for which data were taken, and diameter growth at SEPAC average 0.4 inches per year. These differences in management and inter-tree competition may explain why the heritability for diameter and height for the seedlings at SEPAC continued to increase from age 10 through age 20. Lack of spatial replication of the SEPAC plantation and its management methods probably inflated the heritability estimates derived at the site. But the argument for later selection, at least at sites like SEPAC where growth was excellent, is based not on the absolute magnitude of the heritability estimates, but on the timing of the cessation of rank shifting and the observed increases in the heritability of height and diameter after age 10. The large intra-block variation and a confusing planting scheme at the MSU study site makes family identification and analysis of the growth data from these trees difficult (Paul Bloese, personal communication) and greatly complicates comparisons with the trees at SEPAC and Pleasant Valley.

Tissue Culture Propagation

In order to propagate elite selections and to genetically modify fine hardwoods, we have developed several tissue culture systems. Embryogenic cultures of northern red oak and black walnut were established using cotyledon tissue explants. Cultures have been maintained on maintenance medium and upon transfer to maturation medium, somatic embryos can be matured and germinated. Germinated somatic emblings have successfully been transferred to soil. In addition, shoot cultures of black walnut were established from coppice shoots of greenhouse-grown trees. Recently, shoots have been successfully rooted and transferred to soil. Acclimatization studies for rooted microshoots and somatic emblings will be initiated in the near future.

Vegetative Propagation

A project is underway to determine the conditions necessary for successful cutting propagation of black walnut. Vegetative propagation will be required to produce clones of genotypes selected for improved wood quality, growth, and pest resistance. Successful propagation of black walnut on a commercial scale may be achieved if the type of cutting (hardwood versus softwood), date of collection (growth stage of development), rooting treatment (auxin type and concentration), and greenhouse parameters (mist bed, supplemental lighting, etc.) are carefully considered. Based upon successful propagation methods established for butternut (*J. cinerea*), the effects of the aforementioned parameters on rooting percentage, number and length of roots regenerated, and over-wintering survival of rooted cuttings will be determined. Plants transplanted to the field will be monitored for growth and survival.

CONCLUSION

HTIRC, guided by its Advisory Committee that consists of resource professionals, University and federal scientists, directors of industry and

landowner associations, and research administrators, will seek to engage in collaborative hardwood research and development across the Midwest. Areas of opportunity include breeding and mass selection, genomics and gene discovery, seed and nursery production systems, hardwood silviculture, and technology transfer.

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DESIGNING SPECIFIC CHLOROPLAST MARKERS FOR BLACK WALNUT FROM A SET OF UNIVERSAL PRIMERS

Erin Victory, Rodney L. Robichaud, and Keith Woeste[1]

ABSTRACT.—Chloroplasts are a valuable source of genetic information because their sequence is highly conserved, they undergo little or no recombination, and they are uniparentally inherited. Chloroplast polymorphisms are powerful genetic tools for identifying matrilineal family groups, studying gene flow from seed versus pollen movement, reconstructing phylogeographic colonization, and for elucidating reproductive patterns in natural stands. Here, we sequenced six samples representing the range of black walnut (*Juglans nigra* L.) using six previously published polymorphic chloroplast markers. These markers were from the atpF intron, 3' to rps2, ORF 77-ORF 82 intergenic, ORF 74b-psbB intergenic, and rp12-rps19 intergenic regions in the chloroplast genome. Data from Weising and Gardner (1999) show short mononucleotide repeats of A, T, and G, both individually and in tandem depending on locus. Preliminary data suggest that in black walnut, polymorphisms in these regions are based on series of A and T repeats, and for some loci indicate considerable differences from all the species previously analyzed. New primers specific for black walnut will be designed based on these polymorphisms. These new primers will be used in a study of genetic diversity across the Central Hardwood region.

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DIVERSITY AND RELATEDNESS IN A BLACK WALNUT SEED ORCHARD

Keith Woeste and Doug Mersman[1]

ABSTRACT.—Geneticists and silviculturists have selected over 450 black walnut clones for inclusion in the black walnut breeding program at Purdue University over the past 35 years. Most of the selections were from Indiana; a few were from other states in the Central Hardwoods Region. Selection of second and third generation clones out of this founder population was based primarily on timber traits, such as straightness and diameter growth. While maintenance of genetic diversity was considered a priority, there were no good mechanisms to quantify the presence or loss of diversity and the accumulation of inbreeding in open-pollinated populations that were not isolated from the pollen of unselected trees. Thirty-four superior clones from the Purdue walnut breeding program were evaluated for levels of inbreeding and allelic diversity using eight independently segregating dinucleotide microsatellite markers. The loci used were highly polymorphic in the 24 studied clones, with and average of 11 alleles per locus. The inbreeding coefficient (F) of the population of clones was over four times that found in a typical wild population (0.39 vs. 0.08), indicating that the superior trees may have been the product of close matings or even self-pollinations. The first-generation, mass-selected clones in the breeding program had significantly greater heterozygosity than second-generation clones, and both sub-populations from the program had significantly more homozygosity than wild trees. Surprisingly, despite their relatively high levels of homozygosity, the clones in the breeding program had greater allelic richness (one measure of genetic diversity) than a stand of native-grown trees of comparable size.

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EFFECTS OF VESICULAR-ARBUSCULAR MYCORRHIZAE AND SEED SOURCE ON NURSERY-GROWN BLACK WALNUT SEEDLINGS

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ABSTRACT.—A nursery study was established in Missouri to evaluate the effects of endomycorrhizal inoculation and seed source on the growth of black walnut seedlings. Inoculation, in general, resulted in seedlings with significantly larger sturdiness quotients. *Glomus intraradicies* was found to produce larger seedlings than *Glomus etunicatus*, but only differences in height were significant. Seed source affected the quality of seedlings and produced highly significant differences in height and number of first order lateral roots. Inoculation by seed source interactions was significant only for height growth. However, the study clearly demonstrates variation in walnut growth responses with changes in endomycorrhizal symbiont and seed source. Preliminary results indicate that improvement in black walnut seedling quality can occur as a result of inoculating some walnut genotypes with specific endomycorrhizal organisms.

Black walnut (*Juglans nigra* L.) is a valuable timber species that grows extensively throughout the eastern United States. The species is site demanding exhibiting superior growth on deep, fertile, well-drained soils (Fowells 1965). Walnut is managed for both wood and nut production (Garrett and Harper 1999), although trees that exhibit both superior wood and nut production characteristics are uncommon (Rink 1985).

Although black walnut tree improvement began in the early 1960s (Bey 1970), it has recently intensified with the establishment of the Hardwood Tree Improvement and Regeneration Center at Purdue University and a new emphasis on walnut breeding within the University of Missouri Center for Agroforestry (Tourjee and others 1999). Black walnut has also attracted a great deal of "cultural" research in an effort to expedite the initial growth response following outplanting while increasing overall growth and survival.

The effects of vesicular-arbuscular mycorrhizae (VAM) on black walnut have not been extensively studied. However, Melichar and others (1986), Kormanik (1985), Kormanik and others (1982), among others have published research on the effects of VA mycorrhizae on black walnut growth. The effect of inoculating a variety of black walnut seed sources with different mycorrhizae symbionts and monitoring the walnut genotype/endomycorrhizal interaction in a nursery setting has had limited examination and is the purpose of this work.

The specific objectives of this study were to

1) examine nursery growth responses including height, diameter, and number of first order lateral roots > 2 mm of 11 black walnut half-sib families to three endomycorrhizal treatments, and
2) determine the significance of the black walnut genotype by *Glomus* genotype interaction in the nursery, especially its effect on the rankings of families among the endomycorrhizal treatments.

STUDY AREA AND DESIGN

The study was installed at the Missouri Department of Conservation, George O. White State forest nursery. The nursery is located approximately 40 km south of Rolla, MO, near Licking, MO. The soil is predominantly Huntington silt loam and contains approximately 45 percent silt, 40 percent sand, and 15 percent clay.

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A randomized split-block design was used in this study. Main plots were endomycorrhizal treatments (3), and sub-plots were black walnut half-sib families (11). Thirty-three treatments, with 45 nuts planted to each treatment, were replicated three times for a total of 4,455 nuts planted.

Three endomycorrhizal treatments,
1) *Glomus intraradicies* (Schenck),
2) *G. etunicatus* (Becker & Gerd.), and
3) noninoculated (sterilized sand) were used in this experiment.

Both mycorrhizal symbionts were obtained from Nutri-Link Inc. (Salt Lake City, UT), and shipped in a sand carrier to the University of Missouri. Upon arrival, the inoculum was placed in cold storage until used. The sand for the control treatment was purchased locally and sterilized before used.

Sub-plot treatments consisted of 11 black walnut half-sib families collected from trees in a progeny test located on Hammon's Products Company land near Stockton, MO (table 1). After collection, nuts were hulled, soaked 24 h in water, treated with 5 percent Captan to surface disinfect, placed in large plastic bags according to family, and stored at 2º C to stratify until planting.

The experiment was established at the nursery in late April. One seedbed approximately 85 m in length was utilized for planting. Seedbeds on each side of the experiment were left fallow, and all three beds were fumigated approximately 3 weeks prior to inoculation and seeding. Each mycorrhizal treatment was assigned at random to one of three main plots in each replication. A 3-m buffer zone was left between inoculum treatments. The inoculum was spread evenly at a rate of 1 L per 6.7 m^2. The inoculum was then tilled into the seedbed to a depth of approximately 15 cm.

Table 1.—*Origins of the parent trees of the 11 half-sib black walnut families utilized in the experiment*

Family ID #	Parent origin
02	G.O.W. Super Seedling[1]
03	Shannon County, MO
04	G.O.W. Super Seedling
05	G.O.W. Super Seedling
06	G.O.W. Nursery Run Seedling[2]
07	G.O.W. Nursery Run Seedling
08	G.O.W. Super Seedling
09	Polk County, MO
10	Cole County, MO
11	Cole County, MO
12	Boone County, MO

[1]Large caliper seedlings (> 1.25 cm) of unknown origin taken from the George O. White (G.O.W.) State forest nursery and planted near Stockton, MO in April 1976.
[2]Nursery run seedlings (< 1.25 cm caliper) taken from the George O. White (G.O.W.) State forest nursery and planted near Stockton, MO in April 1976.

Seed sources were randomly assigned to planting locations within each main plot. Nuts of a particular seed source were placed in the seedbed on a 10.2 x 10.2 cm spacing. A 0.6 x 1.2 m plywood template with drilled holes was used to control spacing. Buffers were not used between seed sources thus allowing spacing comparable to normal operating procedures. Seed sources were separated by a lathe to ensure seed source identification. Each seed source was represented by 45 nuts (5 rows of 9 each) within each main plot. After the sixth week of growth, seedlings were fertilized weekly for 6 weeks with foliar-applied urea at a rate of 13.44 kg/ha.

MEASUREMENTS AND ANALYSES

Seedlings were lifted conventionally in December and measured immediately. Height, diameter, and number of first order lateral roots (> 2 mm in diameter) were recorded for each seedling. A random sample of two seedlings from each treatment in each replication was used to determine the intensity of endomycorrhizal colonization, root volume, root weight, and top weight. Seedlings were severed at the root collar and the root system and top placed in separate bags. Roots were cleared of host cytoplasm using 10 percent KOH, alkaline H_2O_2, and 1 percent HCl then stained with 0.01 percent acid fuchsin-lactic acid solution for mycorrhizal assessment (Kormanik and others 1981).

A 0.5- x 0.5-cm grid was prepared and placed in the bottom of a standard petri dish in which cleared root segments were randomly placed. At each point where a root segment intersected a grid line, a determination was made as to whether or not mycorrhizal structures (either hyphae, vesicles, or arbuscules) were present. One hundred intersection points were observed for mycorrhizal structures. This procedure provides a direct relationship to percentage of lateral roots colonized for a particular seedling.

Seedling tops were dried at 85º C for 48 h and weighed. The root system was divided

vertically into two equal parts and each half was immersed in a volumetric cylinder to determine water displacement (Burdett 1979). All components of the root system were re-combined, dried at 85° C for 48 h and weighed.

All data were subjected to analysis of variance using the SAS system for personal computers (SAS Institute, Cary, NC). Treatment degrees of freedom were partitioned to perform orthogonal contrasts to compare various treatment combinations (Neter and others 1996). Orthogonal contrasts were used to compare endomycorhizal treatment means (main plot effect) of the measured seedling attributes. Two degrees of freedom allowed for two planned contrasts. The first contrast compared the effect of inoculation versus no inoculation (control), and the second compared the effect of inoculation with *G. intraradicies* versus inoculation with *G. etunicatus*.

RESULTS

Inoculation produced seedlings with a significantly greater ($p < 0.05$) sturdiness quotient than control seedlings (table 2). Sturdiness quotient measures the stocky or spindly condition of seedlings (Thompson 1985), with a high quotient indicating more spindly seedlings.

While not statistically significant ($0.05 < p < 0.10$), inoculation also yielded considerably higher values for height growth, stem weight, colonization percentage, and root volume than noninoculation (table 3). Height growth was the only measurement for which *G. intraradicies* inoculated seedlings was better than *G. etunicatus* inoculated seedlings. With the exception of shoot: root ratio, *G. intraradicies* inoculated seedlings had larger mean values for all measured attributes than either *G. etunicatus* inoculated or control seedlings.

The effect of half-sib families (sub-plot effect) on both height growth (table 4) and number of first order lateral roots > 2 mm (table 5) was highly significant ($p < 0.01$). Although family rankings varied with the measurement taken, Family 09 consistently ranked in the top five in all attributes measured with the exception of height, while some families (e.g., 07) ranked consistently low (table 6).

The inoculation treatment by family interaction was statistically significant for height growth ($p < 0.05$) indicating rank and magnitude changes of families in height growth response by inoculation treatment. This interaction was not statistically significant for any of the other traits measured. However, the rankings of the families by each of the treatments were inconsistent within and across all traits measured.

DISCUSSION

A nursery study was established to evaluate the significance of inoculating black walnut seed sources with two species of endomycorrhizal-forming fungi in improving seedling quality. *Glomus intraradicies* and *Glomus etunicatus*

Table 2.—*Black walnut family sturdiness quotient[1] means and overall treatment and family means of inoculated and noninoculated nursery-grown seedlings*

Family	Mycorrhizal treatment and associated sturdiness quotients			Family mean
	Noninoc.	*G. intraradicies*	*G. etunicatus*	
	Quotients			
02	3.30	4.66	4.94	4.30
03	5.39	7.49	6.62	6.50
04	5.46	5.27	5.91	5.55
05	5.77	6.07	6.40	6.08
06	4.78	5.13	4.41	4.77
07	3.83	6.69	5.58	5.37
08	5.75	6.04	4.46	5.42
09	5.76	4.32	5.33	5.14
10	4.37	3.87	5.79	4.68
11	5.06	6.21	5.08	5.45
12	4.56	7.13	4.88	5.52
Treatment means:	4.91[2]	5.72	5.40	5.34

[1]Sturdiness quotient is ht. (cm) of the seedling divided by the stem diameter (mm).
[2]Orthogonal contrasts found a significant difference ($P = 0.02$) between treatment means of noninoculated vs. inoculated seedlings. No other significant differences were detected with ANOVA.

Table 3.—*Mean values of all seedling attributes measured in the nursery. Families were subjected to inoculation with* Glomus intraradicies (GI), Glomus etunicatus (GE) *and sterilized sand (no inoculation (NI)).*

Seedling Attribute	Noninoculated	*Glomus intraradicies*	*Glomus etunicatus*	Grand mean
Height (cm)	44.6	45.8	49.6[1]	46.7
Stem dry weight (g)	5.65	6.51	8.24	6.80
First order lats > 2 mm	3.4	3.6	3.7	3.5
Percentage colonization	33.2	40.9	55.1	43.1
Root dry weight (g)	15.3	19.4	20.5	18.4
Shoot:Root ratio	.473	.432	.452	.452
Root volume (ml)	47.6	55.6	59.1	54.1
Caliper (mm)	8.46	8.67	8.86	8.66
Sturdiness quotient	4.9	5.4	5.7[2]	5.3

[1]Orthogonal contrasts indicated that G.I. inoculated seedlings had heights that were significantly ($0.05 < P < 0.10$) greater than G.E. inoculated and noninoculated seedlings.
[2]Orthogonal contrasts indicated that inoculation resulted in seedlings that had significantly ($P = 0.02$) greater sturdiness quotients than noninoculated ones.

Table 4.—*Black walnut family height means and overall treatment and family means of inoculated and noninoculated nursery-grown seedlings*

Family	Mycorrhizal treatment and associated mean heights			
	Noninoc.	*G. intraradices*	*G. etunicatus*	Family mean
	cm			
02	45.02	42.25	44.79	44.02[2] bcde
03	51.44	52.42	44.96	49.61 abcd
04	40.13	47.84	52.53	46.83 abcde
05	51.00	54.89	44.98	50.29 ab
06	42.62	51.64	40.95	45.07 abcde
07	44.29	51.58	50.57	48.81 abcd
08	53.47	52.26	43.51	50.52 a
09	45.40	41.44	44.48	43.69 cde
10	33.18	42.32	49.56	41.69 e
11	45.30	58.28	46.16	49.92 abc
12	38.91	50.75	40.47	43.38 de
Treatment means:	44.61[1]	49.61	45.83	46.68

[1]Orthogonal contrasts indicated no significant differences among mycorrhizal treatments.
[2]Means sharing the same letter within provenance means are not significantly different at $P < 0.05$ using Duncan's New Multiple Range Test.

were utilized. Attempts to improve seedling quality through cultural treatments have not led to an agreement on parameter(s) that best correlate with outplanting success. Williams and others (1985), working with walnut reported that geographic seed origin was best correlated with field performance of nursery stock. Thompson (1985), working with conifers, measured a number of seedling parameters that can be used as predictors of outplanting success. These parameters are either direct seedling measurements or composites of two or more variables. One measurement, sturdiness quotient, was used in our study to determine its' correlation with black walnut outplanting success. Because there is not just one or two seedling parameters that have been demonstrated to be good indicators of field performance in walnut, many other variables were also evaluated.

Table 5.—*Black walnut family first order lateral root means and overall treatment and family means of inoculated and noninoculated nursery-grown seedlings*

Family	Mycorrhizal treatments and associated mean first order lateral roots > 2 mm.			
	Noninoc.	*G. intraradicies*	*G. etunicatus*	Family mean
	number of laterals > 2 mm			
02	2.54	3.59	2.33	2.82[2] f
03	3.07	2.23	2.53	2.61 g
04	2.76	2.74	2.04	2.51 g
05	4.37	3.43	3.15	3.65 de
06	3.35	3.48	4.18	3.67 d
07	2.02	1.95	2.28	2.08 h
08	3.66	5.56	7.02	5.21 b
09	5.01	5.10	6.64	5.45 a
10	3.50	6.23	1.89	3.87 c
11	3.42	2.77	4.51	3.57 e
12	3.32	3.16	4.89	3.79 cd
Treatment means:	3.36[1]	3.66	3.57	3.53

[1]Orthogonal contrasts indicated no significant differences among mycorrhizal treatments.
[2]Means sharing the same letter within provenance means are not significantly different at $P<0.05$ using Duncan's New Multiple Range Test.

Table 6.—Rankings[1] of family means for seedling parameters measured in the nursery

Rank	Root vol.	Ht.	Dia.	Lats > 2mm	Stm. Wt.	Rt. Wt.	% Colon.
	Family						
1	05	08	09	09	05	09	05
2	09	05	08	08	04	11	12
3	11	11	06	10	11	08	08
4	10	03	11	12	08	05	09
5	02	07	02	06	09	02	06
6	08	04	05	05	03	04	10
7	03	06	10	11	02	10	03
8	06	02	03	02	10	03	11
9	04	09	12	03	12	06	07
10	12	12	04	04	07	12	02
11	07	10	07	07	06	07	04

[1]All rankings are from highest to lowest means.

Inoculation Effects

A significant response ($p < 0.05$) to mycorrhizae inoculation was observed in sturdiness quotient and near-significant differences ($0.05 < p < 0.1$) were found for height growth, stem weight, percentage colonization, and root volume (table 3). Of the two organisms evaluated, *Glomus intraradicies* proved superior to *Glomus etunicatus* when all growth indices were evaluated collectively.

Generally, other studies have shown the same positive effects of inoculation on walnut seedling morphological characteristics that we observed (Schultz and others 1981, Schultz and Kormanik 1982, Kormanik 1985, Melichar and others 1986, Dixon 1988). As in the present study, inoculated seedlings have been found to be larger than their nonmycorrhizal counterparts. Moreover, in each of the studies that have compared the effect of different species of

Glomus, each has shown that no single *Glomus* species consistently produces the largest seedling.

Height, caliper, and number of first order lateral roots > 1 mm and 2 mm are morphological characteristics that are most often measured in the nursery. Also, these characteristics are most often used to correlate nursery stock with outplanting success and subsequent survival and growth. For example, Williams (1965) concluded that black walnut seedlings 0.64 cm or larger in diameter survive and grow best following outplanting.

In the present study and in Dixon's work (1988), a significant or near-significant height response resulted from mycorrhizal inoculation. Dixon (1988) utilized *G. etunicatus* and found it to positively stimulate height growth of seedlings. In studies conducted by Melichar and others (1986), Kormanik (1985), and Schultz and Kormanik (1982) no changes in height resulted from inoculation with various strains of endomycorrhizae. However, none of these researchers utilized either *G. intraradicies* or *G. etunicatus*. The significant height response reported by Dixon (1988) and found in the present study, but not found by others, would seem to indicate the importance of the mycorrhizal isolate/walnut genotype interaction.

Changes in height will affect the sturdiness quotient of seedlings when caliper is minimally changed. Because height was significantly affected by inoculation in our study but not caliper (the denominator), inoculation resulted in seedlings with significantly higher sturdiness quotients than was observed without inoculation. Sturdiness quotient has not been included in other black walnut seedling studies; therefore, the optimal quotient is unknown. Moreover, the range of sturdiness quotients (high or low) that signifies good quality seedlings is unknown for black walnut.

Caliper response to mycorrhizal inoculation was not significant in our study. However, *G. intraradicies* inoculated seedlings were 4.5 percent larger in caliper than the controls. Moreover, in a follow-up study, inoculation with *G. intraradices* resulted in seedlings that were 10.6 percent larger in diameter than the noninoculated controls (data unpublished). Some variation in caliper response to inoculation also exists in the literature. Kormanik and others (1982) indicated that black walnut seedlings inoculated with *Glomus fasiculatus* and a mixture of VAM fungi were both 27.8 percent larger in diameter than the controls. Dixon (1988) and Melichar and others (1986) have also reported variations in seedling diameter response depending on the endomycorrhizal organism evaluated.

Although not statistically significant, inoculation resulted in a 7.1 percent increase in the number of first order lateral roots > 2 mm. Variation in lateral root promotion with inoculation has also been reported in the literature. Melichar and others (1986) reported that inoculation with a mixture of *G. microcarpus* and *G. fasiculatus* resulted in a significant increase in lateral root numbers. However, inoculating with *G. microcarpus*, *G. mosseae* or *G. caledonius* did not. Dixon (1988) reported significantly more lateral roots promoted with inoculation, but no differences existed among *Glomus* treatments. Dixon (1988) also found lateral root promotion dependent on walnut genotype.

Stem weight and percentage colonization were affected by inoculation but not significantly. As in the present study, Kormanik (1985) reported that mycorrhizal inoculation had an effect on stem weight and percentage colonization ($0.05 < p < 0.10$). Overall comparisons were not reported by Melichar and others (1986), but significant differences in stem dry weight and percentage colonization did exist among treatments. It is noteworthy that the replication by treatment interaction (error a) approached significance ($p < 0.06$) for percentage colonization in our study suggesting that much unexplained variation was present for percentage colonization. This factor alone may explain why there were no significant differences in percentage colonization found between inoculation treatments.

As is evident from this discussion, the response of walnut to endomycorrhizal inoculation as reported in the literature is highly variable. Even changing sites within a nursery and the corresponding changes in soils along with the changes in weather from year-to-year may serve to increase the variation observed. Kormanik (1985) emphasized the role that phosphorous (P) plays in mycorrhizae research. As P availability increases beyond a threshold value in the soil, the effect of mycorrhizal inoculation on seedling growth responses may decrease. Bray 1 P tests in our study revealed an average of approximately 162 kg of P per ha, which is usually not high enough to inhibit mycorrhizal development. However, variation observed within the treatment bed could account for some of the

differences observed. Sword and Garrett (1994), working with an ectomycorrhizal species, found levels of boron to have an effect on colonization. Black walnut genotype may also have been a factor in the varying response of mycorrhizal organisms.

Seed Source Effects

Seed source was found to be significant for height and number of lateral roots > 2 mm. Dixon (1988) found that provenance was significant for root collar diameter, root weight, leaf area, total root system length, and VAM colonization in his study. Other authors have also demonstrated the importance of provenance selection (Bey 1980, Williams and others 1985).

In our study, only one provenance consistently ranked in the top five for the variables measured, emphasizing the importance of selection in securing fast growing planting stock. Family 09 from a select tree located in Polk County, MO, consistently ranked in the top five for all parameters measured with the exception of height (table 6). Family 09 exhibited a height that was 6.4 percent less than the overall provenance mean. However, this family was 6.1 percent larger in caliper and had 35.3 percent more lateral roots > 2 mm than the overall mean. Moreover, it had consistently higher values than the overall means in all other growth attributes measured. Similarly, Dixon (1988) found that one provenance (out of three tested) ranked first in caliper and lateral root promotion (> 2 mm) and second in height regardless of treatment. Family 04, from a George O. White Nursery super seedling planted near Stockton, MO, consistently ranked near the bottom in all measured parameters.

Treatment * Family Interaction

Height was the only growth parameter that was found to have a significant treatment * family interaction. Dixon (1988) studied the effects of mycorrhizal inoculation on growth and development of three black walnut seed sources. He found that the seed source/mycorrhizal interaction was significant for root collar diameter, root weight, leaf area, total root system length, and percentage colonization. None of the interactions described by Dixon (1988) were found to be significant in our study.

Interactions demonstrate that there are rank changes or changes in the magnitude of response of families to different inoculation treatments. To illustrate this point, families 07, 11, and 12 when inoculated with *G. intraradicies*, produced seedlings that were taller than the controls. However, inoculation of families 02 and 09 with *G. intraradicies* resulted in seedlings shorter than the control.

Dixon (1988) evaluated three organisms, *G. margarita*, *G. etunicatus*, and *G. deserticola*, of which only *G. etunicatus* was tested in our study. In contrast to our findings, *G. etunicatus* performed well in Dixon's (1988) greenhouse experiment. However, the response to inoculation depended on the seed source. Dixon's work and the results of our study clearly demonstrate variation in growth response in black walnut with changes in endomycorrhizal symbiont and seed source.

CONCLUSIONS

The following conclusions can be drawn from this study:

1) Family growth rankings can change as a result of endomycorrhizal colonization.
2) Caution must be exerted when making walnut nursery selections because the best family growth may be a result of a treatment/genotype interaction and not superior genetics alone.
3) Further research is needed to determine the value of optimal black walnut/endomycorrhizal symbiont combinations.
4) Preliminary results indicate that improvement in black walnut seedling quality can occur as a result of inoculating some walnut genotypes with specific endomycorrhizal organisms.

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SURVIVAL AND GROWTH OF DEEP-PLANTED, IN-LEAF GRAFTS IN A GERMPLASM REPOSITORY OF CANKER-RESISTANT BUTTERNUT

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Butternut (*Juglans cinerea* L.), highly valued for its timber and nuts, occurs as widely scattered trees or isolated stands throughout the Central Hardwood region (Rink 1990). The introduced fungus *Sirococcus clavigignenti-juglandacearum* has rapidly cankered, girdled, and killed most of the butternut trees; however, a few trees that may be resistant to the fungus have been found (Furnier and others 1999). A program to graft scionwood from putatively canker-resistant trees and establish germ-plasm repositories within the natural range of butternut has been initiated as one of several conservation measures (Ostry and others 1994, 2002). Although butternut easily grafts to seedling rootstocks of black walnut (*J. nigra* L.) and heartnut (*J. ailanthifolia* Carr.), the overwintering of containerized grafts has been problematic.

The objectives of our study were

1) to evaluate the feasibility of mid-summer field-planting of in-leaf grafts,
2) to establish a germplasm repository in the southern part of the natural range, and
3) to test for differences in virulence of the pathogen as found in the Central Hardwood region.

Black walnut seedlings to be used as rootstocks were obtained from the Minnesota state tree nursery and Forrest Keeling Nursery in Elsberry, MO, and planted into 10-cm-square x 30-cm-tall plastic pots. In March 1994, scionwood from 31 putatively canker-resistant butternuts from Minnesota, Wisconsin, New York, and Connecticut were side grafted on the basal stem of the rootstocks (table 1). In mid-June, successful grafts were moved to open shadehouses. In mid-July 1994, in-leaf grafts were transported from Minnesota to the Tree Improvement Center near Carbondale, IL, which is located 10 km west of the Southern Illinois University-Carbondale campus.

Approximately, 2 and 6 weeks before establishing the planting, the area was broadcast sprayed with glyphosate to deaden the tall fescue sod. Planting holes, 60- to 75-cm deep on a 2.5- x 2.5-m spacing, were prepared with a tractor-mounted 20-cm-wide posthole digger in relatively moist Hosmer silt loam soil with a 45- to 60-cm-deep fragipan. To minimize any "clay pot" effects, the packed sidewalls of planting holes were cut with a tile spade and dropped into the bottom of each hole. Grafts were planted using soil from the borings with the rootstock root collar near the soil surface or deeper. Two to five grafts of each clone were randomly assigned to each block of eight rows of 13 trees.

Landscape fabric, 60-cm-square, was placed around each graft and covered with a 10-cm-deep layer of pine/hardwood mulch for weed control. Weed control in 1995 consisted of cultivation between the rows and spot application of glyphosate and simazine in 1996 and 1997. To protect grafts from rabbit and deer damage, a 20-cm-diameter x 75-cm-tall cylinder of 1.2-cm-square hardware cloth was placed over each graft and anchored with a bamboo pole. A wire stake with a stamped aluminum tag identified each graft as to clone number.

Overhead irrigation was provided for 24 to 48 hours after planting and weekly thereafter so that the precipitation averaged 2.5 cm per week. Many grafts settled deeply into the planting holes with many having graft unions below the soil line. In June and July 1995, basal sprouts on rootstocks were removed, but were allowed to grow thereafter if the scion had died. Survival of rootstock and scion, presence of cankers,

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Table 1.—*Clone number; ortet location by state and county, diameter at breast height (cm), appearance of cankers on putatively resistant trees; number of grafts planted; second, fourth, and seventh year survival of grafts on live rootstocks; and seventh year height, diameter at ankle height, and stems on butternut grafts in the southern Illinois germplasm repository*

Clone	Ortet location		Ortet	Cankers	Total	Surviving grafts on live rootstocks			Seventh Year[a]		
no.	State	County	DBH	present	Grafts	Yr 2	Yr 4	Yr 7	Ht.	DAH	Stems
			-cm-		-no-	-%-	-%-	-%-	-m -	-cm-	-#-
1	WI	Dunn	41.7	Maybe	16	50	14	14	2.6	5.0	1.0
5	MN	Rice	36.8	No	8	43	29	29	1.5	2.0	2.0
6	MN	Houston	44.2	No	16	21	11	0	xxx	xxx	xxx
20	WI	Rock	33.5	No	20	67	33	36	2.6	5.0	1.5
21	WI	Rock	35.1	No	12	27	17	17	1.9	4.3	4.0
22	WI	Rock	34.5	No	8	43	43	43	3.0	6.4	1.5
23	WI	Rock	32.3	No	8	50	50	60	1.6	2.2	1.6
29	WI	Pepin	41.9	One	16	33	22	12	2.3	4.4	1.0
32	WI	Marquette	30.5	No	12	45	0	0	xxx	xxx	xxx
34	WI	Pepin	46.0	Maybe	16	44	11	11	1.8	2.8	2.0
35	WI	Pepin	18.0	Small	16	54	38	38	1.4	3.4	2.0
36	WI	Burnett	31.0	Few	20	18	10	11	1.4	2.0	1.0
37	WI	Burnett	33.0	Branch	20	56	27	27	1.6	2.9	2.0
38	WI	Burnett	18.8	No	12	17	13	14	1.0	1.4	3.0
39	WI	Burnett	17.5	No	12	36	13	13	1.4	4.8	2.0
50	WI	Price	42.4	No	20	59	42	33	1.7	3.2	1.9
51	WI	Forest	46.7	No	20	50	30	20	1.0	2.1	1.2
52	WI	Forest	63.5	No	12	67	56	38	1.6	3.7	2.0
55	WI	Forest	46.1	No	12	50	0	0	xxx	xxx	xxx
56	WI	Langlade	36.8	No	16	38	13	14	1.2	3.3	2.0
57	WI	Langlade	36.3	Yes	8	17	0	0	xxx	xxx	xxx
61	WI	Rock	28.4	No	20	47	40	40	2.2	4.8	2.1
63	WI	Rock	29.2	No	8	29	14	0	xxx	xxx	xx
64	MN	Wabasha	44.5	One	8	57	25	33	2.2	6.3	1.5
69	MN	Hennepin	35.1	No	16	63	53	40	1.8	3.0	1.4
70	WI	Burnett	38.6	No	12	33	14	14	2.0	4.8	2.5
71	WI	Burnett	38.9	No	8	43	20	20	3.3	7.8	1.5
95	NY	Jefferson	30.2	No	8	50	50	50	xxx	xxx	xxx
98	NY	Jefferson	41.9	No	12	73	22	11	1.7	3.0	1.5
101	CT	Hartford	>30	No	16	64	60	60	3.3	6.8	1.6
104	CT	Hartford	>30	No	8	83	80	80	2.8	7.8	2.7

[a]Height, diameter, and number of stems does not include grafts repeatedly mowed off along the north border of the planting.

height, and diameter at ankle height were determined in fall of 1995, 1996, 1997, and 2001. In 1995 and 1997, we determined the origin of live stems relative to the soil surface. The morphology of dormant terminal buds was used to determine species in 1997 and 2001.

Survival after the second growing season of deep planted walnut rootstocks (graft union more than 10 cm deep) with and without live butternut scions was less than survival of walnut rootstocks with graft unions less than 10 cm deep or above the soil surface (table 2).

Survival differences of walnut at different planting depths were no longer significant after the fourth growing season. Williams (1974) previously reported that deep planting of black walnut had no effect of seedling survival.

After two growing season, survival of the black walnut rootstocks ranged from 50 to 100 percent among the 31 clones. Walnut rootstock survival decreased slightly through the seventh growing season when it varied from 42 to 94 percent among the clones. Half of the repository was located in a draw that frequently had standing

water in the spring or after irrigation. Survival of walnut rootstocks in the draw averaged 43 percent compared to 95 percent for rootstocks on a south-facing slope and ridge in the other half of the repository. Presumably, the deep planting holes cut through the fragipan produced unacceptable growing conditions for the black walnut rootstocks considered intolerant of poorly drained soils.

Butternut scion mortality after two growing seasons was higher on surviving deep-planted walnut rootstock than on rootstocks with graft union slightly buried or still above the soil line (table 2). Scions continued to be lost through the fourth growing season; however, differences in mortality no longer existed for planting depth of the walnut rootstock, which can partially be explained by too few surviving butternut grafts.

After two growing seasons, butternut scion mortality on the live walnut rootstocks ranged from 17 to 83 percent among the 31 clones (table 1). After four growing seasons, scion survival among clones ranged from 0 to 80 percent. Only 8 of 27 clones showed additional mortality through the seventh growing season.

Although source trees for some clones had cankers, no cankers have been observed on the butternut saplings in the repository. A few grafts show repeated annual shoot die-back and sprouting from unknown causes. Five grafts exhibit visual symptoms of witch's broom, which is severely affecting some butternut hybrids in an adjacent planting. In one of the four blocks, the ambrosia beetle (*Xyloborus germanus*) has caused dieback of several butternut stems and resprouting of black walnut rootstock; however most attacks have been on black walnut saplings that were allowed to develop after the scion died.

After seven growing seasons, deep planted walnut rootstock on which the scion had died had more rapid basal sprout growth than sprout growth on walnut rootstocks where the grafting scar was above the soil line or slightly buried (table 3). Likewise, butternut scions on deep planted walnut rootstocks had greater shoot growth than successful butternut grafts where the graft union was buried less than 10 cm deep or above the soil line. Overall, height growth of butternut scions was more rapid than walnut sprout growth on failed grafts. This is consistent with previous reports that butternut has more rapid stem growth than black walnut (Schroeder 1972). Many of the butternut grafts had multiple stems in contrast to the sprouted black walnut rootstock that had self-thinned in most cases to a single stem.

After seven growing seasons, height varied greatly among the 27 surviving clones ranging from 1.0 to more than 3.3 m (table 1). Likewise, diameter growth also varied greatly among the 27 clones ranging from 2.0 to 7.8 cm in diameter-at-ankle height. Too few grafts survived for most clones for any statistical comparisons among clones. Scionwood from the more vigorous clones in the Illinois germplasm repository was recently harvested for inclusion in a new repository being established by the Hardwood Tree Improvement and Regeneration Center in northern Indiana.

In summary, we did establish a germplasm repository of putatively canker-resistant butternut clones. The high mortality of the walnut rootstocks and of butternut scions on surviving

Table 2.—*Survival of black walnut rootstocks and butternut scions on live walnut rootstocks*

Species	Scion location[1]	Survival[2] Yr 2	Yr 4	Yr 7
		%		
Walnut	Above	88 a	71 a	67 a
	0 to 10	88 a	74 a	71 a
	>10 cm	81 b	71 a	70 a
Butternut	Above	57 a	36 a	35 a
	0 to 10	48 ab	34 a	29 a
	> 10 cm	39 b	21 a	21 a

[1]Location of graft union or scar above soil line, 0 to 10 cm deep, or deeper than 10 cm.
[2]Means with column and species followed by different letters are different at p = 0.05 percent.

Table 3.—*Stem height for black walnut rootstocks without live scions and for butternut scions grafted on live walnut rootstocks*

Species	Scion location[1]	Height[1] Yr 2	Yr 4	Yr 7
		m		
Walnut	Above	0.4 a	0.7 a	1.4 b
	0 to 10	0.4 a	0.7 a	1.3 b
	>10 cm	0.5 a	0.8 a	1.5 a
Butternut	Above	0.4 a	0.7 a	1.6 c
	0 to 10	0.4 a	0.8 a	2.0 b
	> 10 cm	0.5 a	0.9 a	2.3 a

[1] Means with column and species followed by different letters are different at p = 0.05 percent.

walnut rootstocks suggests mid-summer field-planting of in-leaf grafts with irrigation is not a good alternative to the traditional overwintering of grafts in coolers and subsequent spring planting. Moist soil conditions of the poorly drained soils in this respository, however, can explain the high mortality of the walnut rootstocks in the years following planting. High failure rates for butternut grafts on walnut rootstocks has also been observed in repositories in Vermont and Wisconsin that were established with spring planting of dormant stock (Ostry and Moore 2001). Because no genetic variation has been found within fungal isolates from cankered butternut (Furnier and others 1999), we should not expect to see changes in virulence to the pathogen among the putatively resistant clones used in this repository.

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DENDROCHEMICAL RESPONSE TO SOIL FERTILIZATION

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ABSTRACT.—Use of chemical element content of tree rings to detect soil acid or base changes was tested at 13 sites of former forest fertilization trials in the eastern United States and Canada. Ammonium sulfate or nitrogen fertilization was the typical acidification treatment, while lime added with or without other fertilizer was the typical base treatment. Molar ratios of calcium/manganese and magnesium/manganese in bolewood were used to detect changes in tree ring tissue from 10 hardwood and 3 conifer species available at these sites. Base fertilization at all sites caused molar ratios to increase in all species tested and acid treatment caused ratios in tree rings to decrease. Diffuse-porous hardwood species and conifers with wide sapwood regions were not useful in preserving a chemical record of soil changes caused by fertilization. Approximate chronologies of soil chemical change due to fertilization were approximately preserved in tree cores from Japanese larch and in ring-porous hardwoods sampled up to 30 years after fertilization. Greater use of dendrochemical sampling is recommended to assess fertilizer and soil acidification effects on trees.

Dendrochemistry, or the study of the chemical element content of tree rings, has the potential for detecting and reconstructing environmental changes that influence soil chemistry over the life of the tree. The major limitation of the dendrochemical method for detecting soil chemical change has been the lack of controlled studies of changing soil environments and incomplete understanding of radial translocation and storage of elements in xylem. Natural variations in the availability of soil nutrients and trace metals with soil pH have been found in several tree species (DeWalle and others 1991, Guyette and others 1992, Mohamed and others 1997), but the responsiveness of tree-ring chemistry to soil chemical changes has been documented in only a limited number of studies and species (McClenahen and Vimmerstedt 1993, Kashuba-Hockenberry and DeWalle 1994, DeWalle and others 1999, Watmough and Hutchinson 1996).

Bondietti and others (1990) proposed use of molar ratios of aluminum (Al)/calcium (Ca) or Al/magnesium (Mg) in tree-rings as indicators of change. Studies by Cote and Camire (1995) and Mohamed and others (1997) showed relationships between available soil aluminum and tree-ring aluminum. DeWalle and others (1999) found Ca/Mn and Mg/Mn ratios in bolewood to be useful in studying impacts of soil acidification, while Watmough and Hutchinson (1996) showed Ca, Mg, Mn, and strontium (Sr) were responsive to surface soil pH changes. Shortle and others (1997) provided evidence that cation levels in tree-rings of red spruce (*Picea rubens* Sarg.) increased due to base cation mobilization induced by atmospheric deposition. Concepts of ecosystem acidification from atmospheric deposition with initial base cation mobilization followed by base cation depletion are well established in the literature (Aber and others 1989).

In this paper we combine past dendrochemical results largely related to effects of ecosystem acidification with data from other past fertilization trials where other types of soil base (liming)

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and acid fertilization occurred. The objectives of these studies were to:

1) demonstrate the responsiveness of tree-ring chemistry in a large number of tree species to the effects of both soil acidification and base addition conditions and
2) to assess the ability of tree rings to preserve a record of past soil chemical changes even where up to 30 years had elapsed since the original soil fertilization.

METHODS

Samples of bolewood from treated and control trees for chemical analysis were collected at a number of sites of past fertilization experiments summarized in table 1. Fertilization occurred several years to over 30 years prior to sampling. In most treatments the fertilizers were applied once to the soil surface in granular or powdered form without incorporation into the soil; however, treatments to whole watersheds at Bear Brook in Maine (site 7 in table 1) and Fernow and Clover Run in West Virginia (sites 5 and 6, table 1) were applied several times annually. Fertilization with ammonium sulfate, aluminum sulfate, NPK without lime, and nitrogen-only fertilizers were categorized as acid treatments. Treatments with lime and other base additions, either separately or in combination with NPK and other compounds were classified as base treatments.

In most instances, wood cores were collected from trees (bark to pith) using increment borers (table 1). Black locust (*Robinia pseudoacacia* L.) samples at Fernow (6d in table 1) were taken from approximately 5-cm thick wood discs cut at breast height from felled trees. Only healthy, overstory dominant or co-dominant trees were sampled, although for red maple (*Acer rubrum* L.) and black locust at Fernow (6c and 6d in table 1), only intermediate or suppressed canopy-position trees were available for sampling.

Two types of sampling with increment borers were conducted depending upon availability of trees and other constraints. In some areas, four cores were removed from each of five trees of each species in each treated or control area, while in other areas two or four cores were removed from 10 trees per species. Cores were extracted at breast height in different quadrants of the bole cross-section at all sites except one, which was sampled at stump height (site 4 in table 1). Wood samples were collected with clean, teflon-coated increment corers, handled only with gloved hands, placed in plastic straws and frozen until they were processed.

In the laboratory, cores were separated into 4- or 5-year growth increments to provide enough samples for analysis by inductively coupled plasma emission spectroscopy. Low-power magnification, clean gloves, and scalpels were used for core separation. Growth increments from the same years for each core were composites by tree for chemical analysis. Black locust wood samples, collected from each annual growth increment using a chisel, were combined by year for all treatment or control trees.

Most chemical analyses of the chemical element content of the wood were conducted at the Plant and Soil Analysis Laboratory on the Pennsylvania State University campus, but in some of the earliest sampling other contract laboratories were used. Although a suite of chemical element concentrations were obtained for the wood samples, the molar ratios Ca/Mn and Mg/Mn are stressed in this paper based on past research (DeWalle and others 1999). Data reported are from wood formed just after the last treatments were applied at each study site.

ACID VS. BASE RESPONSE

Dendrochemical responses to acid and base fertilizations were consistent for most tree species (figs. 1 and 2). Molar ratios of Ca/Mn and Mg/Mn in growth rings formed after the most recent fertilization generally showed increases for base fertilization (above 1:1 line) and decreases for acid fertilizations (below 1:1 line). High ratios in base fertilizations were caused by the combined effects of higher concentrations of Ca or Mg and lower concentrations of Mn. Conversely, lowered ratios in acid fertilizations were due to the combined effects of lower Ca or Mg concentrations and increased concentrations of Mn.

The highest Ca/Mn and Mg/Mn ratios for base fertilized trees relative to control trees occurred for red oak (*Q. rubra* L.) at the Laurel Run Acid Mine Site in Pennsylvania (site 12 in table 1), which received the greatest lime application of 28 tonnes ha^{-1} of pulverized limestone. Heavy liming at Cherry Springs in Pennsylvania (site 8) showed nearly equivalent increases in Ca/Mn and Mg/Mn in sugar maple (*Acer saccharum* Marsh.) and black cherry (*Prunus serotina* Ehrh.) trees. In acid fertilized experiments the largest differences between treated and control trees occurred in Japanese larch (*Larix leptolepis* Sieb. and Zucc.) at Clover Run (site 5) and yellow-poplar (*Liriodendron tulipifera* L.), red maple, black cherry, and black locust trees at the Fernow site in West Virginia (site 6).

Table 1.—*Description of forest fertilization sites and denrochemistry studies*

Study and Ref. No.*	Species	Treatment	Amount, kg/ha	Year Treated	Year Meas.	Site	Trees per treat.	Cores per Tree
1 Acid	*Quercus rubra*	Ammonium nitrate	336 N	1977-78	1995	AR, Boston Mtns.	10	4
2 Base		NPK+ lime	1345 N			MO,	7-10	4
2a	*Q. velutina,*		587 P	1973	1995	Sinkin		
2b	*Pinus echinata*		477 K	1973	1995	Exp. For.		
			657 Ca					
			129 Mg					
3 Acid		NPK	448 N			WV,	5	4
3a	*Q. rubra,*		194 P	1970-71	1994	McGee		
3b	*Liriodendron tulipifera*		93 K	1970-71	1994	Run, Mt. Zion Rd:		
4 Acid	*Q. coccinea*	Aluminum sulfate	32.4 S	1988	1994	PA, State College	5	4
5 Acid		Ammonium	35 N			WV,	5	4
5a	*Larix leptolepis,*	sulfate	40 S per year	1987	1994	Clover Run		
5b	*Q. rubra,*			1987	1992			
5c	*Prunus serotina*			1987	1992			
6 Acid		Ammonium	35 N			WV,	5	4
6a	*P. serotina,*	sulfate	40 S per year	1989	1996	Fernow		
6b	*L. tulipifera,*			1989	1996			
6c	*Acer rubrum*			1989	1996			
6d	*Robinia psuedoacacia*			1989	2000		9-10	discs
7 Acid		Ammonium	26.5 N			ME,	5	4
7a	*Acer saccharum,*	sulfate	30 S per year	1989	1994	Bear Brook		
7b	*Fagus grandifolia,*			1989	1994			
7c	*Picea rubens,*			1989	1994			
7d	*Betula allegheniensis*			1989	1994			
8 Base		Liming	11.2 lime			PA,	10	2
8a	*P. serotina*			1985	1994	Cherry		
8b	*A. saccharum*			1985	1994	Springs		
9 Base	*A. saccharum*	Potassium sulfate, lime	224 K 92 S as K_2SO_4, 500 as Ca and Mg CO_3	1989	1994	Quebec, Univ. Montreal, Biol. Sta.	10	2
10 Base	*A. saccharum*	Maple Gro, dried blood, bonemeal, KCl, and dolomite	equiv. to 400 of 4-4-8 fert.	1988	1994	Quebec, Univ. Montreal, Biol. Sta.	5	4
11 Base	*Q. coccinea*	Liming	605 Ca as hydrated lime	1963-64	1990	PA, Black Moshannon	5	4
12 Base	*Q. rubra*	Liming	28000 Ca as ground limestone	1978-80	1988	PA Laurel Run Acid Mine site	10	4

**References: 1 = Snyder and others 1982, 2 = Harris and others 1980, 3 = Auchmoody and Smith 1977, 4 = L. McCormick, Penn State, School of Forest Resources personal communication, 5 = Kochenderfer and others 1995, 6 = DeWalle and others 1999, 7 = Norton and others 1994, 8 = Long and others 1996, 9 = Fyles and others 1994, 10 = B. Cote', U. Montreal personal communication, 11 = Kashuba-Hockenberry and DeWalle 1994, 12 = DeWalle and others 1995.*

Applications of ammonium sulfate several times annually for 8 years appeared to cause the greatest acidification at the Fernow and Clover Run sites.

Some treatments produced ratios that fell closer to the 1:1 line (figs. 1 and 2); differences between treatment and control were not always significantly different. Additions of N fertilizer or NPK fertilizer without lime, such as that in the Boston Mountains of Arkansas (site 1) or the McGee Run (site 3a) and Mt. Zion Road (site 3b) sites in West Virginia, showed evidence of modest acidification response (figs. 1 and 2). Four species at Bear Brook in Maine (site 7) showed modest acid response to treatment with ammonium sulfate treatment. However, unlike trees at Fernow or Clover Run, trees at Bear Brook were sampled after only 6 years of treatment, which was before base cation depletion was developed fully.

Modest base fertilizations of sugar maple in Canada (sites 9 and 10), scarlet oak (*Quercus coccinea* Muenchh.) at Black Moshannon in Pennsylvania (site 11) and shortleaf pine (*Pinus echinata* Mill.), and black oak (*Quercus velutina* Lam.) in Missouri (site 2) showed expected base fertilizer responses, but only modest to small departures from the 1:1 lines (figs. 1 and 2). Aside from the Bear Brook site, modest changes in bolewood chemistry to fertilization in these studies was probably related to the single applications and/or low amounts of fertilizer applied.

Scarlet oak at the State College site in Pennsylvania (site 4) that received aluminum sulfate fertilizer was the only species not exhibiting the expected response for either Ca/Mn or Mg/Mn. Aluminum sulfate was expected to produce an acidifying effect on the soil, but caused a small increase in Ca/Mn or Mg/Mn ratios in tree rings like a base addition. This effect may have been due to replacement of base cations by Al on soil exchange sites and the relatively high base saturation of this soil.

As expected, treatment with nitrogen or NPK fertilizers caused an acidification response in tree-ring chemistry. Liming application with nitrogen fertilizer showed a base response

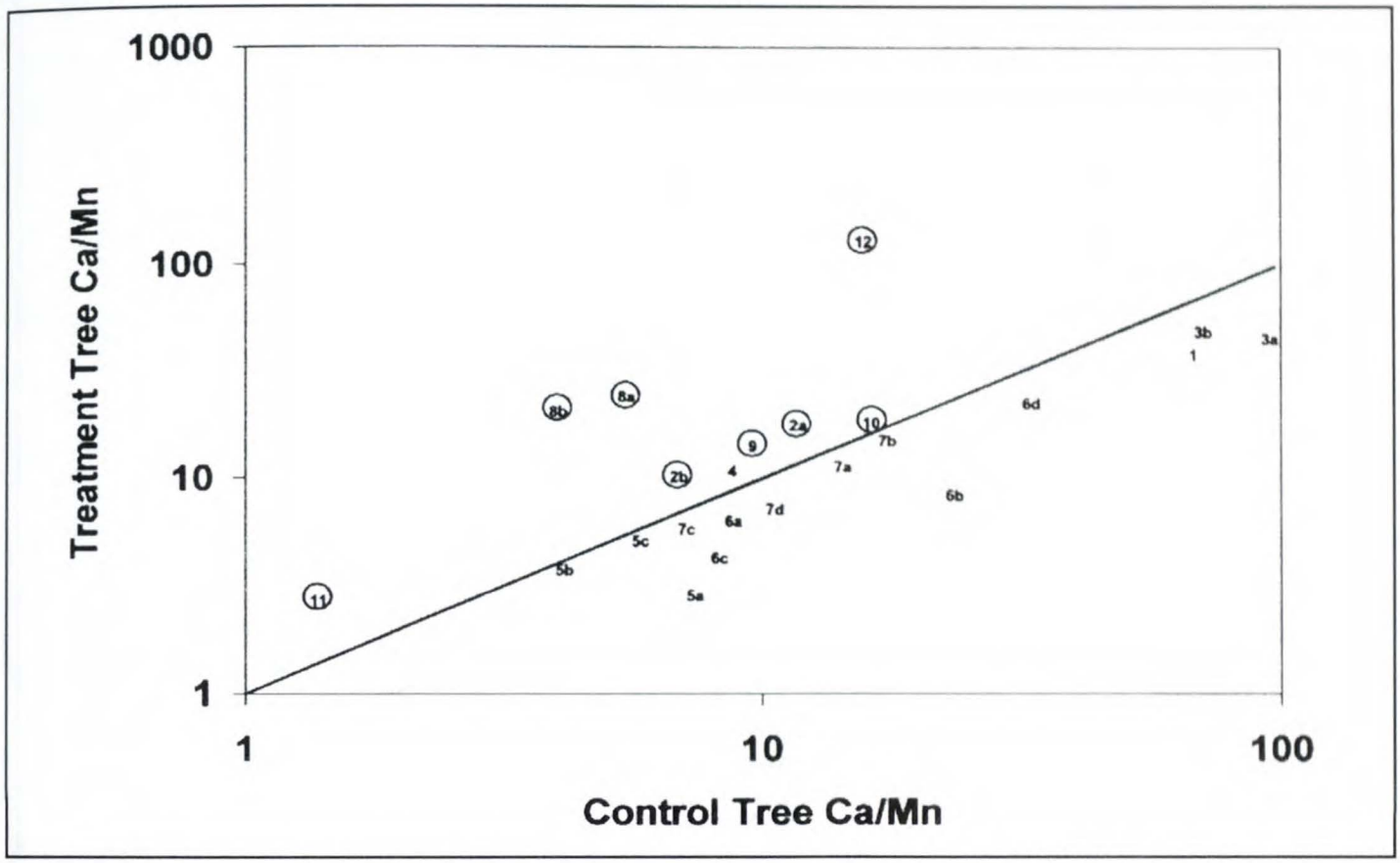

Figure 1.—*Comparison of Ca/Mn molar ratios in bolewood from treated and control trees at sites of forest acid and base fertilization trials. Data were derived from wood growth segments formed after the last treatment. Circles around points denote base cation fertilization, while points without circles denote acidifying fertilization. Numbers and letters of points are keyed to treatments, species, and sites described in table 1.*

indicating that the lime overcame the acidifying effects of N-fertilization.

TREE-RING CHRONOLOGIES

Bolewood sampling to establish the chronology of chemical changes was limited by the width of sap conducting tissue in trees. Trees with wide sapwood tended to show no consistent trends due to fertilization, while those with narrow sapwood showed patterns related to past soil changes. The pattern of Ca/Mn ratios that existed in yellow-poplar cores extracted 30 years after acid treatment with NPK fertilizer at the Mt. Zion site in West Virginia and in scarlet oak cores extracted 25 years after lime treatment at the Black Moshannon site in Pennsylvania are shown for illustration purposes (figs. 3 and 4, respectively). Yellow-poplar is a hardwood species with diffuse-porous wood and a wide band of sapwood, while scarlet oak is a ring-porous hardwood with a moderately wide sapwood zone. The precise width of sapwood and heartwood regions at the time of treatment are not known for either site.

Yellow-poplar showed reduced ratios due to treatment with NPK fertilizer indicating acidification; however, wood tissue formed up to 20 years before and over 30 years after treatment showed effects of treatment (fig. 3). Although yellow-poplar trees appeared quite sensitive to soil chemical change (figs. 1 and 2), the chronology of molar ratios preserved in tree rings shows no evidence of the exact time of treatment. This species clearly cannot be used to date changes in site fertility. Other diffuse-porous hardwood and most conifer species tested, showed similar lack of ability to preserve a chronology of soil chemical change.

Scarlet oak, with its narrower sapwood region than yellow-poplar, showed a different response dendrochemical chronology to lime application (fig. 4). Ratios were similar in treated and control trees prior to treatment. After treatment, relatively high Ca/Mn ratios occurred in rings formed immediately after treatment (1963-1969 segment) and the maximum liming effect appeared about a decade after treatment (1970-1974 segment). The timing of maximum liming

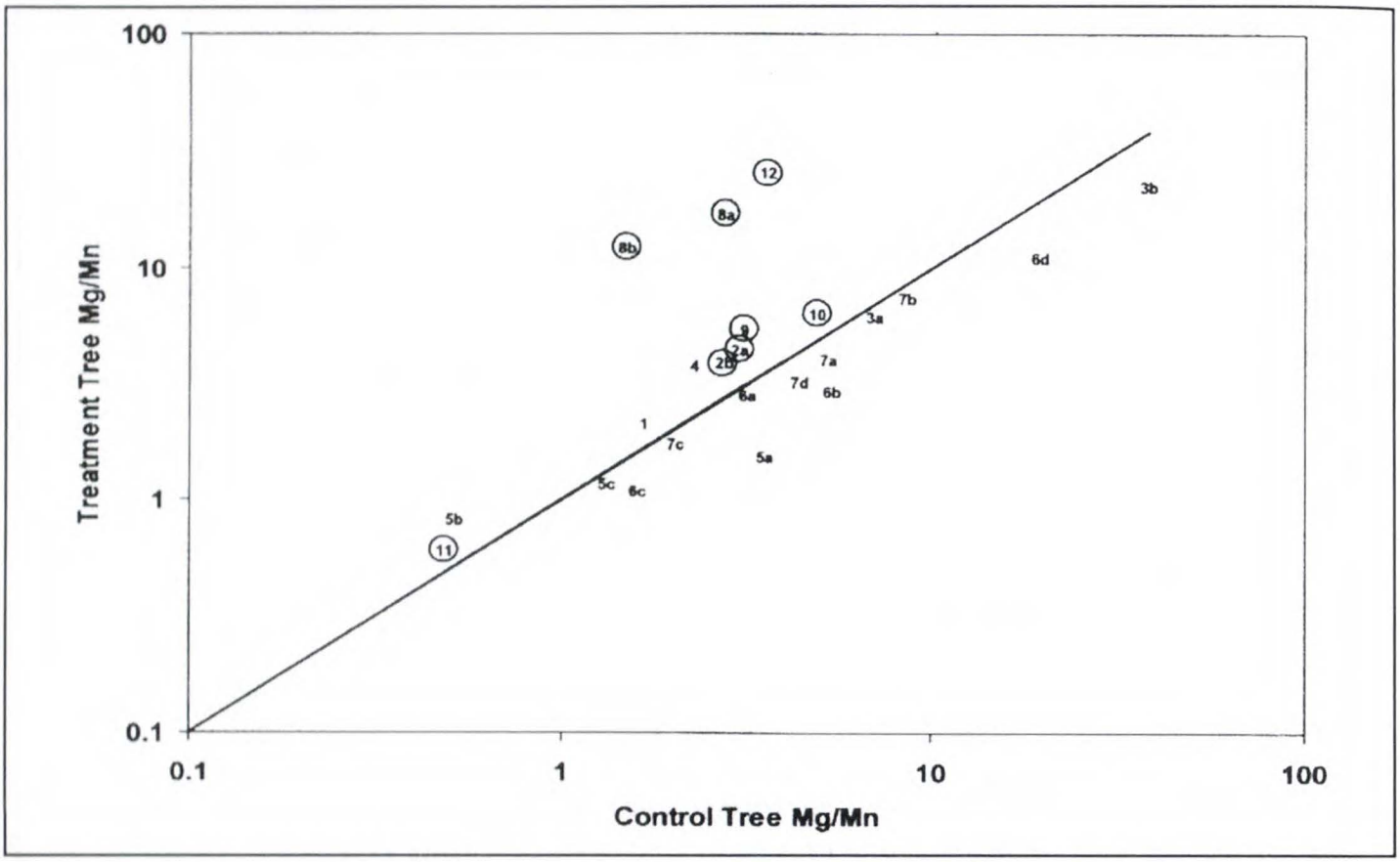

Figure 2.—*Comparison of Mg/Mn molar ratios in bolewood from treated and control trees at sites of forest acid and base fertilization trials. Data were derived from wood growth segments formed after the last treatment. Circles around points denote base cation fertilization, while points without circles denote acidifying fertilization. Numbers and letters of points are keyed to treatments, species, and sites described in table 1.*

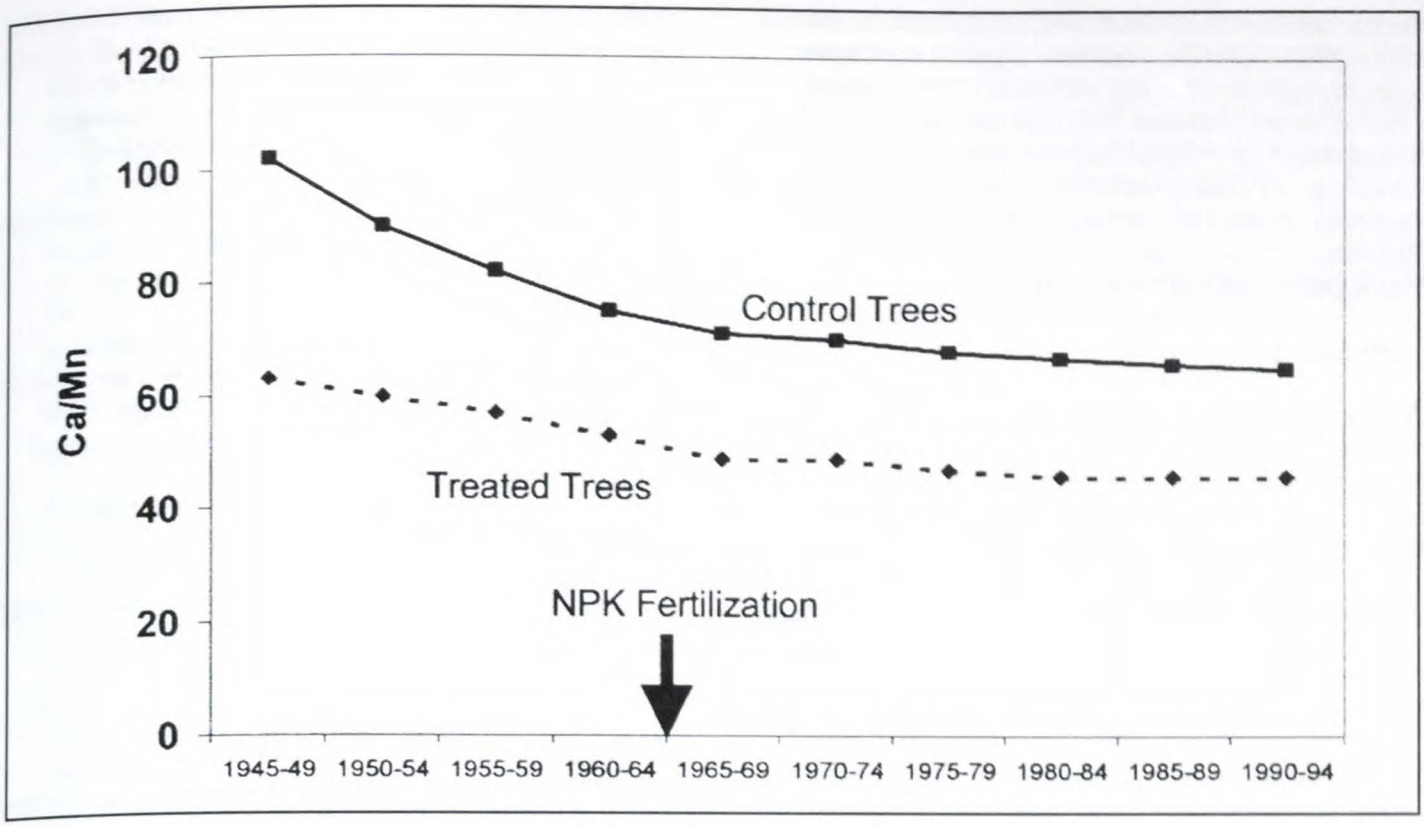

Figure 3.—*Pattern of mean Ca/Mn molar ratios in bolewood from treated and control yellow-poplar trees subjected to one-time NPK fertilization at the Mt. Zion, WV site. Arrow indicates year of NPK fertilization.*

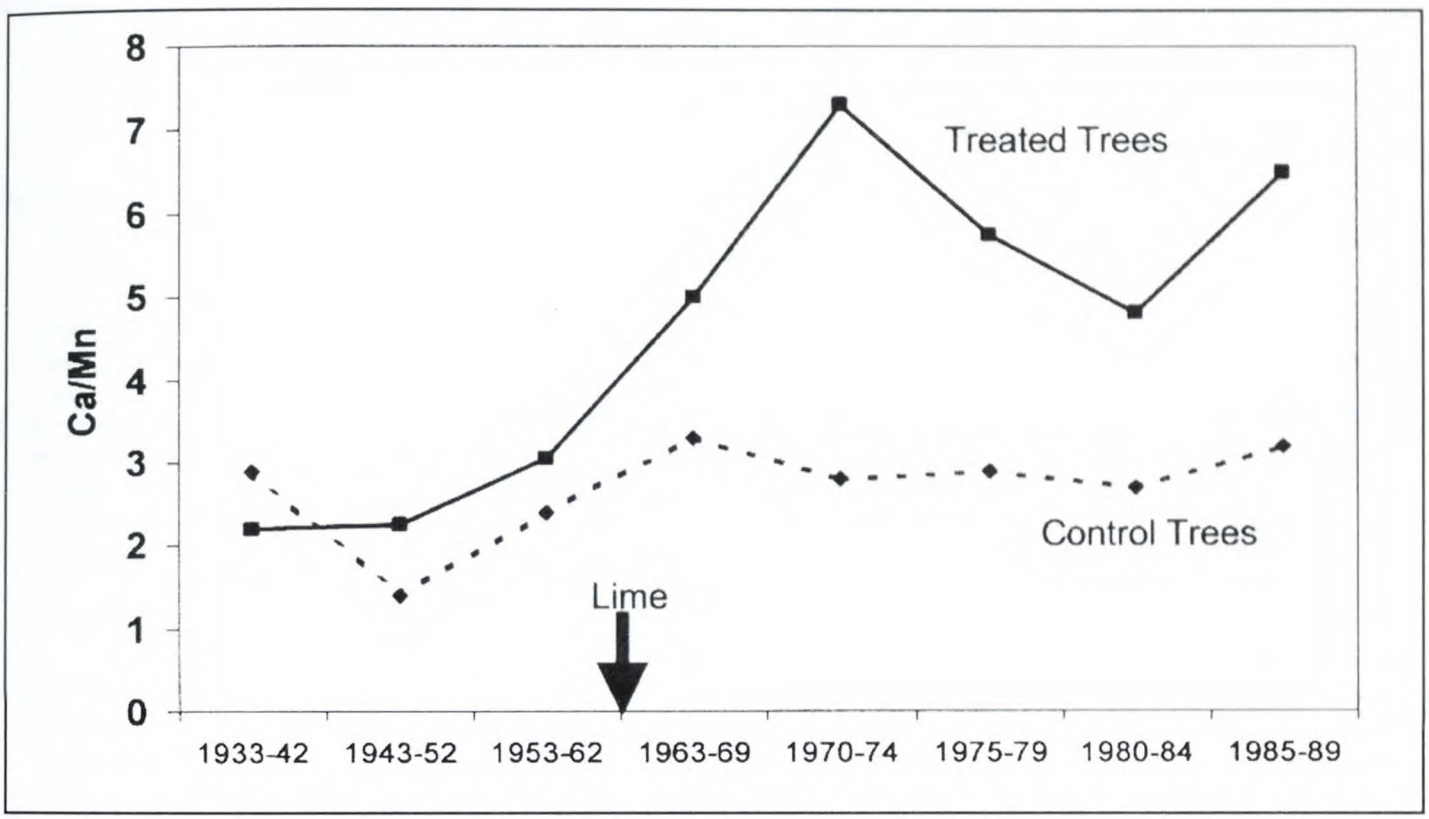

Figure 4.—*Pattern of mean Ca/Mn molar ratios in bolewood from treated and control scarlet oak trees subjected to one-time liming at the Black Moshannon, PA site. Arrow indicates year of liming.*

effect a decade after treatment is possible considering that lime was surface applied and not incorporated into the soil. Gradual convergence of ratios in treated and control trees in wood formed after 1974 was also suggested and this chronology of change after lime treatment was preserved in the bole for up to 25 years after treatment. Thus, it appears that scarlet oak, unlike yellow-poplar, can be used to approximately date changes in soil fertility and plant nutritional status.

All tree species studied were categorized according to their ability to preserve a chronology of chemical changes caused by soil fertilization in tree rings (table 2). Categories are tentative since the magnitude of the treatment and other unknown site conditions also may have influenced the patterns of chemical element changes detected. Since cores were generally divided into segments of 5 years or more, more precise timing of changes could not be determined. The longevity of fertilizer effects on trees is not well known and assessing the effects of forest fertilization could be one of the most useful applications of dendrochemisty.

Japanese larch was the best species tested at providing an exact record of the timing of soil chemical changes. Sapflow in this species appears to be confined to one or two growth rings without much radial translocation of elements (DeWalle and others 1999). Black locust was expected to preserve a good chronology by virtue of its narrow sapwood (4 to 5 rings) and extensive heartwood, but Mg/Mn and Ca/Mn ratios did not show expected trends for unknown reasons.

Table 2.—*Rating of tree species for ability to preserve a dendrochemical chronology*

Species	Chronology preservation
Larix leptolepis	Very good
Quercus rubra	Good
Q. velutina	Good
Q. coccinea	Good
Prunus serotina	Fair
Pinus echinata	Poor
Fagus grandifolia	Poor
Picea rubens	Poor
Betula allegheniensis	Poor
Liriodendron tulipifera	Poor
Acer saccharum	Poor
A. rubrum	Poor

Species with ring-porous wood and narrow sapwood bands, such as the oaks and black cherry, were useful at preserving at least an approximate record of the time of soil chemical changes. Diffuse-porous hardwood species and two other conifers tested, red spruce and shortleaf pine, did not appear to preserve change chronology in any consistent manner.

CONCLUSIONS

Molar ratios Ca/Mn and Mg/Mn in bolewood in a wide variety of deciduous and coniferous species are sensitive indicators of changes in the soil chemical environment due to either acid or base fertilizer treatments. Nitrogen-only, ammonium sulfate, and NPK fertilization of trees without lime caused an acidification response or reduction in Ca/Mn and Mg/Mn ratios in bolewood. Fertilization with lime caused increases in bolewood Ca/Mn and Mg/Mn ratios.

Approximate chronologies of soil fertility changes due to fertilization were preserved for up to 30 years in tree rings of ring-porous hardwood species, such as the oaks and black cherry, and conifers like Japanese larch. However, diffuse-porous hardwoods, black locust, red spruce, and shortleaf pine did not preserve a marked chronological record. Despite limitations imposed by some tree species, careful application of dendrochemistry offers promise as a tool for assessing impacts of fertilization and soil acidity changes.

ACKNOWLEDGMENTS

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BRANCH MORPHOLOGY IN YOUNG POPLAR CLONES ON FLOODPLAIN SITES IN MISSOURI

Stephen G. Pallardy and Daniel E. Gibbins[1]

ABSTRACT.—Four *Populus* clones were grown in central Missouri for 2 years at 1 x 1 m spacing to study total biomass production on floodplain sites previously in forage grasses. Branch morphology (living, first-order proleptic, and sylleptic shoots) was assessed for 2-year-old plants. All 2-year-old plants had lateral branches, and clones varied signi-ficantly in certain branch attributes. A *Populus deltoides* x *P. nigra* hybrid (I45/51) had significantly more branches, greater total branch length and more branches per unit height than did three *P. deltoides* clones (26C6R51, 2059, 1112) derived from Midwest region collections. Further, I45/51 carried a greater proportion of its branches on the lower stem than did *P. deltoides* clones. Whereas intense branching often occurred below 50 cm height in the hybrid clone, *P. deltoides* plants often were clear of branches below 1 to 2 m. Mean angles of branch origin were similar among clones (46.9º to 51.1º) with no significant differences. Length-weighted vector averages of branch azimuth indicated that there was a significant trend toward greater branch growth on the south side of trees, but little apparent clonal variation in this attribute. The profuse branching habit of the hybrid I45/51 was closely associated with its high second-year leaf area index and biomass production.

Populus L. (poplar) species have been incorporated into numerous managed systems for production of timber and fiber. Such culture provides a variety of benefits both economic and environmental. Poplars, and especially their hybrids, have displayed a prodigious capacity for rapid biomass accretion (Anderson and others 1983, Heilman and Stettler 1985, Ranney and others 1987). The rapid growth rates of poplars are related to unit leaf area rates of photosynthesis (which are among the highest of all woody plants (Nelson 1984)), as well as to leaf area production and display. Area-based productivity of poplars depends on planting density and the rate at which canopy closure can be attained. Branch growth and orientation largely govern this process.

Genetic variation in many traits, including growth rate, within species and among hybrids of *Populus* species is well documented. Poplar photosynthesis is under genetic control and sensitive to environmental conditions (Regehr and others 1975, Liu and Dickmann 1993, Rhodenbaugh and Pallardy 1993). Previous work has also shown that poplar clones vary phenotypically in rate and duration of branch and leaf area production and in display characteristics (Isebrands and others 1983, Michael and others 1988, Scarascia-Mugnozza and others 1997). These traits are often highly heritable (Wu 1994, Wu and Stettler 1994, 1996). Hence study of the structure and function of promising clones can aid in selection, tree improvement, and distribution programs.

Despite the popularity of poplar culture and study of poplar biology in other parts of the U.S. (e.g., Pacific Northwest and the Lakes States), there has been little systematic study of the suitability of poplars for use in short-rotation plantations in the lower Midwest, and no study of morphological determinants of growth in genotypes distributed to the public by state agencies. Here we present information on branch morphology, canopy attributes, and growth of four, 2-year-old poplar clones grown in short rotation plantations established in the floodplain of the Missouri River in central Missouri.

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PLANTATION ESTABLISHMENT

Hardwood cuttings of three eastern cottonwood (*Populus deltoides* Bartr.) and one *P. deltoides* x *Populus nigra* L. hybrid clone (table 1) were planted in May 1999 in a former tall fescue (*Festuca arundinacea* Schreb.) pasture on the Missouri River floodplain at the University of Missouri's Horticulture and Agroforestry Research Center at New Franklin, MO (Lat. 39º 01', Long. 92º 46'). Before planting the site was treated with a non-selective herbicide to eliminate grass competition. A randomized complete block design similar to that of Scarascia-Mugnozza and others (1997) was employed, with six replicates (blocks) each consisting of four single-clone plots (7 rows x 10 columns). Spacing was 1 x 1 m (10,000 trees/ha).

Planted cuttings showed good shoot emergence in May, but in June there was substantial mortality in some clones. This required replanting with potted cuttings and transplanting from some blocks so that a number of complete blocks would be available for each clone. Ultimately the number of complete blocks (of an original six) available per clone was: six for I45/51, five for 1112, three for 2059, and two for 26C6R51. Plants were watered frequently during the first growing season to maintain high soil moisture in the top 30 cm, but not during the second year when growing season rainfall (May-September) was adequate (52.3 cm, 102 percent of average).

Height and diameter growth in years one and two were measured for 79 "permanent plot" trees throughout the plantation (17 to 24 trees per clone). Some destructive harvesting was conducted in the plantation to develop biomass prediction equations, but buffer trees were maintained at all adjacent positions around each permanent plot tree.

BRANCH ANALYSES

For the present work, after the second growing season five to six trees per clone were selected for detailed study. In randomly chosen blocks the mean diameter of permanent plot trees was computed and permanent plot trees of approximately this diameter were designated for study. Measurements began at the first branch from the bottom upward. Data obtained were: total tree height and diameter, main stem branch number, branch height above ground, branch length from collar to tip of bud, branch orientation in degrees clockwise from due North, angle of branch origin (acute angle made by the ascending branch to the main stem on the apical side) and terminal leader length. If the lead stem had been cut or damaged, or if two main stems had formed, the dominant stem was considered the main stem and not measured as a branch. Dead branches, which comprised less than 10 percent of total branches, were not measured.

During the second growing season, Leaf Area Index (LAI) and canopy gap estimates were obtained monthly for each plot using a LI-COR LI-2000 Canopy Analyzer (LI-COR, Inc., Lincoln, NE).

Height, stem base diameter, d^2h (a proxy variable for volume), main stem branch number, number of branches per m height, branch length from collar to tip of bud, total branch length per tree, angle of branch, terminal leader length and mid-season (early August) LAI were analyzed via Analysis of Variance. Branch length data were square-root transformed after Nelson and others (1981). Least squares means were computed and means comparisons were conducted (via SAS LSMEANS probabilities) if overall ANOVA results provided a significant F-test ($p < 0.05$).

Table 1.— *Origin and source information for four* Populus *clones grown in a short rotation plantation in central Missouri*

Clone	Parentage	Source	Origin	Latitude	Longitude
I45/51	*P. deltoides*Iowa x *P. nigra*	State Tree Nursery	——	——	——
26C6R51	*P. deltoides*	Missouri Department of Conservation Nursery	Pope County, IL	37º 25'	88º 34'
2059	*P. deltoides*	Missouri Department of Conservation Nursery	Osage County, MO	38º 27'	91º 52'
1112	*P. deltoides*	Missouri Department of Conservation Nursery	New Madrid County, MO	36º 35'	89º 37'

Crown orientation was assessed for all sampled trees by vector averaging (Stull 1995) in two ways, using:

1) unit vectors (branch length = 1 for all branches) and
2) branch length-weighted vector averages.

These data were tested using contingency table analysis for statistically significant trends in N-S and E-W orientation by comparing the number of trees having averaged vectors with some component of N- or E-facing orientation (i.e., 270 to 90º and 0 to 180º, respectively) vs. those with S- and W-facing orientation (90 to 270º and 180 to 360º, respectively). Patterns of branch distribution along the stem were analyzed by goodness-of-fit tests using χ^2 of pooled branch numbers above and below the midpoint between ground level and the highest branch insertion height, with an equal distribution above and below as the null hypothesis.

RESULTS AND DISCUSSION

Plants of all clones typically had excurrent form as shown in figure 1 with lateral branches on lower stems and long terminal leaders, as has been previously reported in *Populus* spp. grown at close spacing (e.g., Dawson and others 1976, Ceulemans and others 1988).

At a finer scale the 2-year-old plants exhibited clonal differences in morphology (table 2). The *Populus deltoides* x *P. nigra* hybrid clones (I45/51) had significantly more ($p \leq 0.05$) branches, greater total branch length, and more branches per unit height than did the three *P. deltoides* clones (26C6R51, 2059, 1112) derived from Midwest region collections.

Whereas intense branching often occurred below 50 cm in height in the hybrid clone, the first live branch of *P. deltoides* clones occurred 1 to 2 m high on the stem. This pattern resulted in significant differences in branch distribution above and below the midpoint of the branching region (i.e., excluding the terminal leader length), with two *P. deltoides* clones (26C6R51, 1112) exhibiting more branches above the midpoint, one clone (2059) with equal distribution, and one clone (I45/51) having branches significantly weighted toward the lower portion of the branching region (table 3).

Nelson and others (1981) also noted development and retention of lower branches in 4-year-old hybrid *Populus* clones grown in short-rotation intensive culture plantations in Wisconsin. The pattern of branching displayed by I45/51 may promote diameter growth, as photosynthate produced by leaves displayed lower in the crown often is preferentially allocated to cambial growth (Larson and Gordon 1969). In fact, mean second-year stem diameter in I45/51 was greater than that of all *P. deltoides* clones (table 4). The diameter differences between clone

Figure 1.—*Appearance of* P. deltoides *x* P. nigra *clone I45/51 (upper) and* P. deltoides *clone 1112 (lower) in November, 2000, after two seasons of growth on a floodplain, formerly-pastured site in central Missouri.*

Table 2.—*Mean values of branch attributes for 2-year-old* Populus *clones grown in a short-rotation plantation in central Missouri*

Clone	Branch angle (º)	Total branch length (m)	Branches/m height	Number of branches	Terminal leader length (m)
I45/51 (n=6)	51.1	22.61 a	6.4 a	30.0 a	1.20
26C6R51 (n=5)	50.2	10.72 b	4.1 b	17.4 b	1.30
2059 (n=6)	46.9	10.25 b	2.5 b	11.8 b	1.78
1112 (n=5	49.2	8.48 b	2.5 b	10.0 b	1.92

[1] Within a column means not followed by the same letter are significantly different ($p < 0.05$).

Table 3.—*Percent of branch insertions above and below the midpoint of total branch insertion regions of the main tree stem for* Populus *clones. Asterisks indicate significant departures from an equal distribution as the null hypothesis by pooled* c^2*; "ns" indicates a lack of significant departure from an equal distribution.*

Clone above midpoint	Percent above midpoint	Percent below midpoint
I45/51	23.5	76.5*
26C6R51	73.6	26.4*
2059	57.7	42.3ns
1112	98.0	2.0*

Table 4.—*Mean values of stem attributes for 2-year-old* Populus *clones grown in a short-rotation plantation in central Missouri*

Clone	Height (m)	Diameter (cm)	d^2h (cm^3 x10^{-2})	Mid-season leaf area index
I45/51 (n=24)	4.67 a	4.67 a	109.7 a	3.98 a
26C6R51 (n=14)	3.58 b	3.52 b	30.0 b	2.37 b
2059 (n=21)	5.19 a	4.15 ab	94.9 ab	2.40 b
1112 (n=20)	4.33 ab	3.67 b	60.2 b	2.26 b

[1] Within a column means not followed by the same letter are significantly different ($p < 0.05$).

I45/51 and clones 1112 and 26C6R51 were statistically significant ($p \leq 0.05$).

Branch distribution between the two height growth increments (HGI) appeared to vary within and among clones (fig. 2). Whereas clone 1112 produced very few to no branches in the second year, clone 26C6R51 had many more first-order branches on the second HGI. Clones 2059 and I45/51 exhibited greater plant-plant variation in HGI distribution patterns and branch distribution on both HGIs. Although I45/51 had some second-year HGI branches, there were far more first-year HGI branches in all plants.

Proportional distribution of branch length along the vertical sequence of branch insertion indicated roughly symmetrical distribution of branch length up the stem in three of four clones (fig. 3). Although there was some plant-plant variation, plots of percent branch length versus percent cumulative branches approximated a 1:1 ratio in clones 2059, 26C6R51 and I45/51. In contrast, in four of five plants of clone 1112, relative branch lengths were longer at upper branch insertion points, with the remaining plant largely adhering to the 1:1 relationship observed for other clones.

Mean vertical angles of branch origin were similar (46.9° to 51.1°) with no significant differences among clones (table 2). These values are in the low- to mid-range of previously-reported branch angles for young, closely spaced *Populus* clones of diverse parentage (Nelson and others 1981, Wu and Stettler 1994). Vector averages of branch azimuth indicated that there was a tendency for net branch growth to be greater on the south side of trees (fig. 4). This trend toward a S-facing orientation was significant ($p \leq 0.05$) for branch length-weighted vectors based on a χ^2 test of goodness-of-fit based on a null hypothesis of even N-S distribution. There was little indication of clonal variation in this attribute (not shown). These data suggest that crown architecture responds to the directional nature of the light regime, placing more foliage on the south side of the crown, which in the Northern Hemisphere receives more direct growing season radiation.

Overall, branch morphology in these *Populus* clones was related to several growth attributes. For example, the profuse branching habit of hybrid I45/51 was closely linked with higher mid-season LAI in this clone and the highest value of d^2h, which is closely related to volume growth (tables 2 and 4). Canopy gap estimates for I45/51 at this time were about 5 percent (data not shown), indicating full canopy closure at 1 x 1 m spacing by the second year of growth. Larson and Gordon (1969) also noted the association of abundant branches lower on the stem with greater diameter growth in this clone.

Data from biomass estimation studies (unpublished) indicated that second-year, above- and below-ground biomass also was highest in clone I45/51 (17.7 mt ha^{-1}). The *P. deltoides* clones had sparser branching characteristics that were reflected in lower LAI values (table 4) and higher canopy gap estimates (data not shown). For clones 1112 and 26C6R51 these morphological patterns were associated with smaller second-year biomass (10.62 and 8.42 mt ha^{-1}, respectively). However, it is interesting to note that clone 2059 exhibited volume and biomass growth closer to that of the hybrid clone (second-year biomass estimates of 16.1 mt ha^{-1}) despite dissimilar canopy characteristics. These data suggest

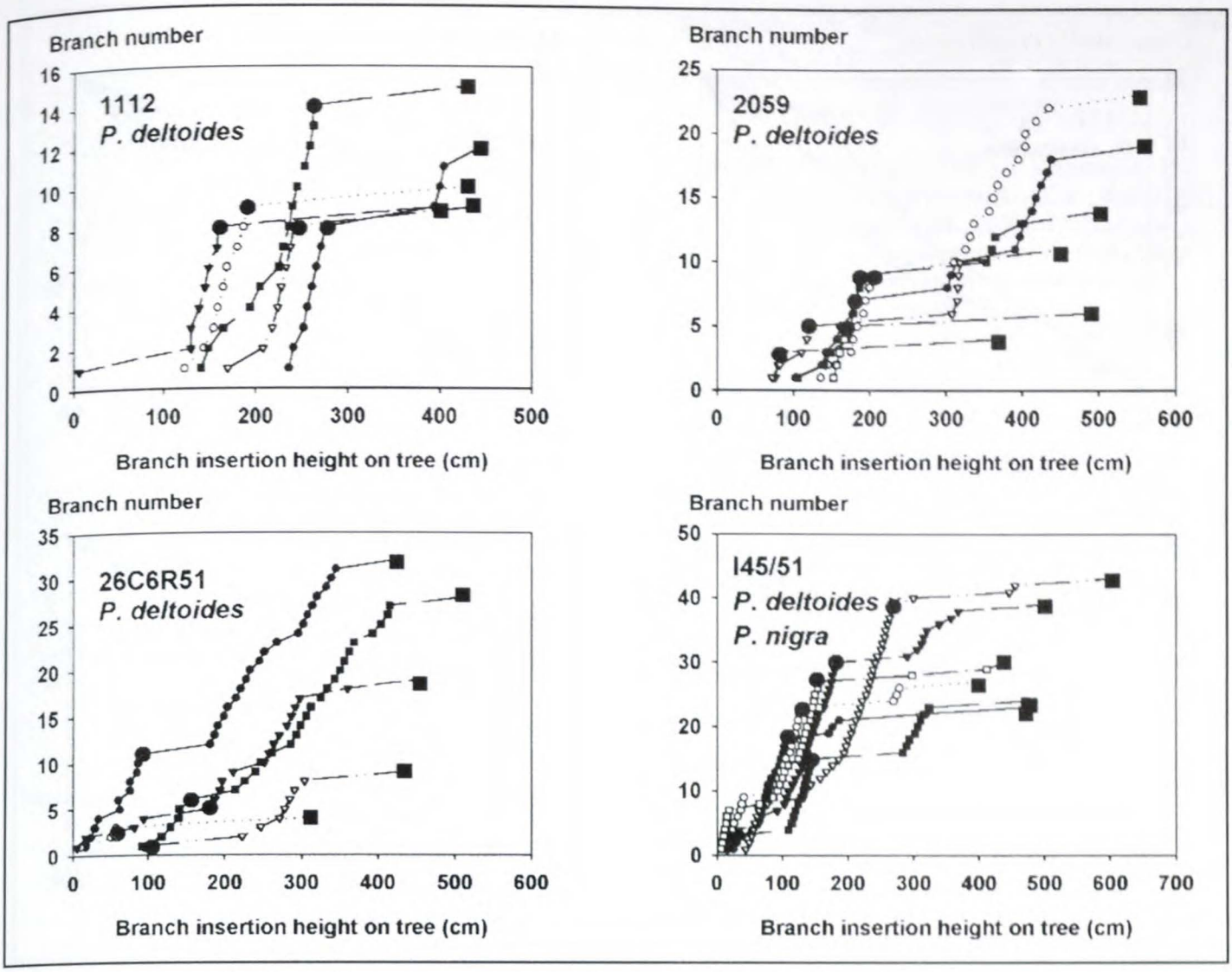

Figure 2.—*Vertical distribution of branches along the stems of four, 2-year-old* Populus *clones. Each trace represents one plant. Large filled circles and squares indicate the limit of first- and second-year height growth increment (HGI), respectively.*

higher photosynthetic capacity and/or assimilation efficiency in clone 2059 than in other clones.

CONCLUSION

Two-year-old clones exhibited significant variations in several branch morphology attributes, primarily relating to branch number, length, and distribution along the main stem. These differences may be partially responsible for associated differences in LAI and growth in plantation culture.

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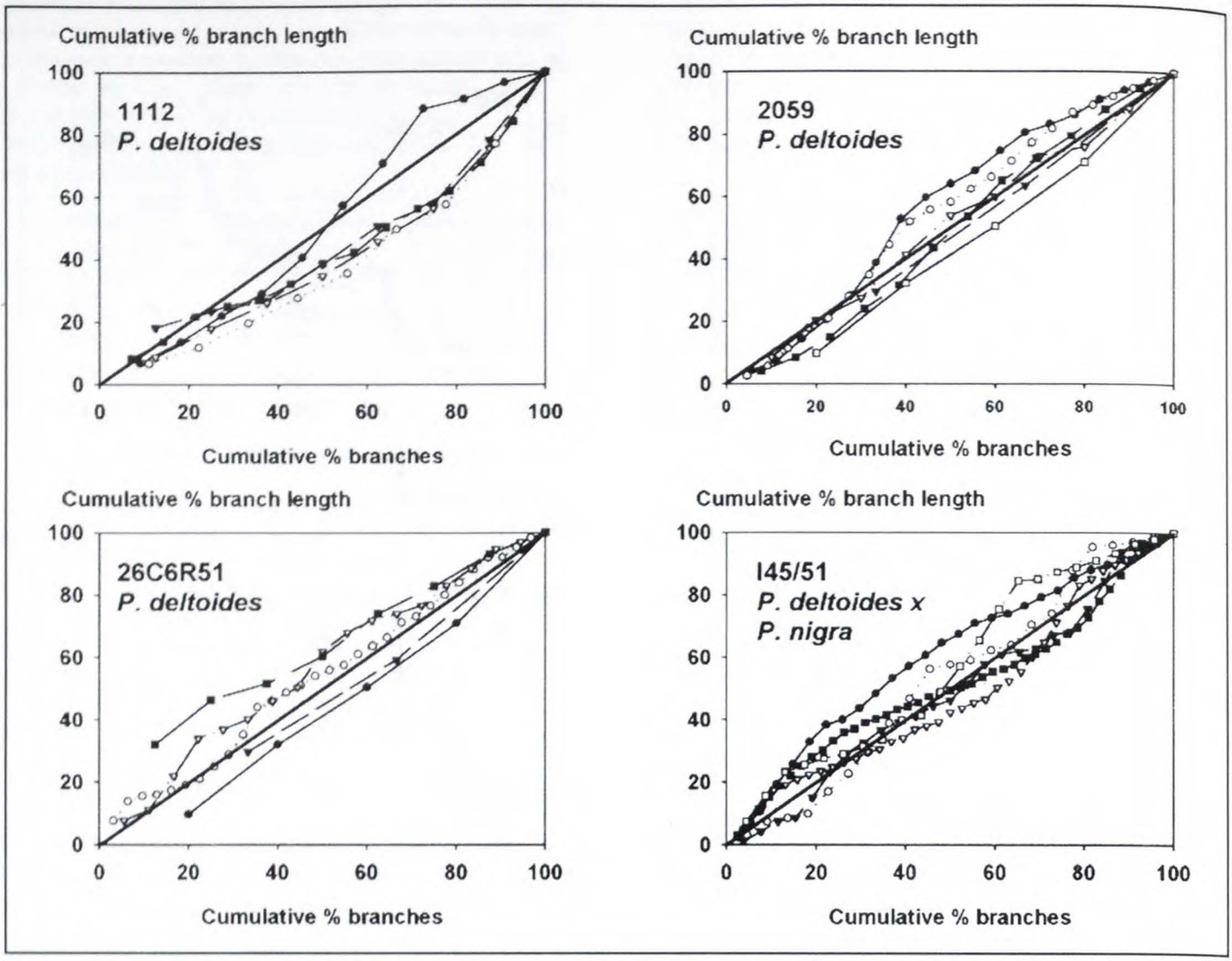

Figure 3.—*Proportional distribution of total branch length (as percent) versus cumulative percent total branches (0 percent = stem base) of four, 2-year-old* Populus *clones. Each trace represents one plant. Heavy line indicates a 1:1 ratio.*

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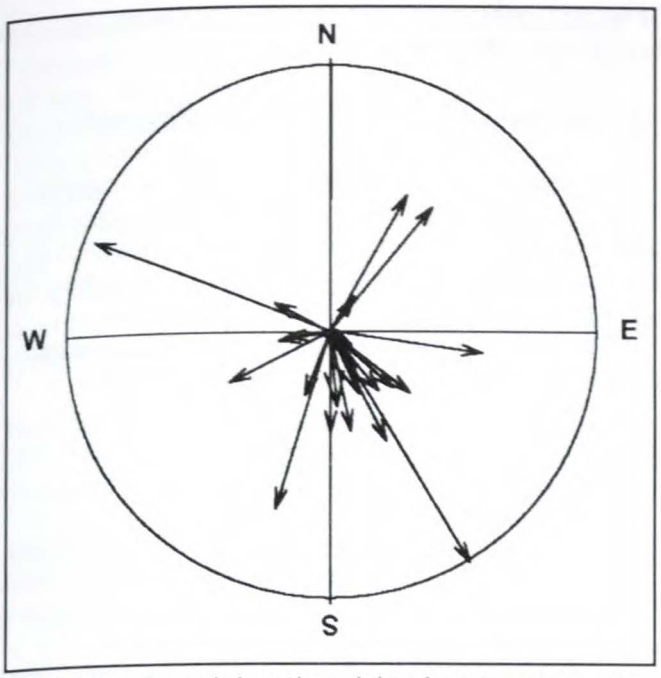

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SAP SUGAR PARAMETERS OF SILVER MAPLE PROVENANCES AND CLONES GROWN ON UPLAND AND BOTTOMLAND SITES

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ABSTRACT.—Sap sugar concentration (SSC), sap volume, and stem diameter were measured for 49 different silver maple clones representing a range-wide collection of 13 provenances within replicated upland and bottomland plantations in southern Illinois during the winter of 2001. For comparison, 42 sugar maple trees were sampled in a local sugarbush. Silver maple SSC averaged 1.51 percent and did not differ among sites. Differences existed among silver maple provenances and clones in SSC and sap volume but performance was site dependent. Selected silver maple clones compared favorably with local sugar maple trees which averaged 1.87 percent SSC.

Production of syrup and sugar from the sap of maple trees originated with Native Americans and was adopted by early European settlers as a source of food and income (Larsson and Jaciw 1967, Koelling and others 1996). Of the 13 native North American maple species, sugar maple (*Acer saccharum* L.) and black maple (*A. nigrum* Michx. f.) are the most preferred for commercial tapping. Their relatively high sap sugar concentration (SSC) and sap flows, which generally extend later into the spring compared to some other species, account for this bias. High sap sugar content is crucial because the majority of annual production costs are attributed to energy required in the sap to syrup concentration process (Huyler 1982).

Sugar maple tree improvement research has been focused on selecting high sap sugar trees and developing successful clonal propagation methods to rapidly multiply superior genotypes (Kriebel and Gabriel 1969, Yawney and Donnelly 1982). Sap sugar concentration of black and sugar maple averages between 2.0 to 2.5 percent (Heiligmann and Winch 1996) but may range from 1.0 to 5.0 percent or more among a local population (Walters 1982) or from 0.7 to 10.8 percent among a range-wide population of forest-grown trees (Gabriel 1972).

Genetic improvement with sugar and black maple has progressed slowly as clonal propagation methods, such as grafting, rooting cuttings, or tissue culture are generally difficult or unreliable (Atkison 1964, Santamour and Cunningham 1964, Dirr and Heuser 1987, Godman and others 1990, Zaczek and others 1997) and sexual maturity is not attained until at least 22 years of age (Godman and others 1990). Additionally, without sugar maple clones to control genotypic variability, it is difficult to clearly determine the environmental factors that influence sap sugar concentration and production.

Silver maple (*A. saccharinum* L.), with a greater natural range than sugar or black maple, is infrequently tapped for maple syrup production. This is largely attributed to its' lower average SSC (1.5 to 2.0 percent) and a shorter collection season due to earlier spring budbreak, compared to sugar and black maple (Walters 1982, Heiligmann and Winch 1996). However, in a Canadian study, silver maple SSC averaged 2.6 percent and ranged from 1.7 to 5.1 percent (Larsson and Jaciw 1967), suggesting that there may be sufficient variability within the species to make significant genetic gains through selection and propagation in a tree improvement program.

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Silver maple characteristics such as an extremely rapid growth rate, successful clonal propagation protocols, and differences in spring budbreak phenology (Kopp and others 1988, Preece and others 1991) make this species a promising candidate for genetic improvement in maple syrup production. Furthermore, clonal silver maple plantings could be used to more clearly understand and model the influences of genetics, environment, and their interaction on sap parameters in maple.

The objective of this study was to determine if there were differences in sap and size parameters among a variety of clones and provenances of silver maple trees grown on contrasting upland and bottomland sites. Additionally, silver maple SSC performance was compared to that of sugar maple.

STUDY AREAS

This research was conducted on replicated plantations of source-identified silver maple provenances and clones previously established in 1990 for biomass production studies (Preece and others 1991, Ashby and others 1993). The plantations are located in southern Illinois; one in a riparian zone (Chautauqua Bottoms) subjected to periodic flooding and one on a drier upland site (Thunderstorm Road). Each plantation was originally planted with 52 clones arising from 13 provenances, (four clones per provenance) planted in three-tree clonal plots randomly located in each of 10 blocks. Trees were planted on 2 m spacing with two border rows surrounding the plantation.

In 1995 all trees were cut to determine above ground biomass and the coppice was allowed to regrow. After coppicing, the trees developed multiple stems. In 1996, at Thunderstorm Road, in each three tree clonal plot within five blocks, coppiced trees were randomly thinned to either a single stem, to two stems, or left unthinned. Trees from five thinned blocks at Thunderstorm Road as well those within five unthinned blocks at Chautauqua Bottoms were utilized in this study of maple sap characteristics.

METHODS

In January 2001, the largest diameter tree in each three-tree clonal plot was chosen for tapping. For multi-stemmed trees the largest stem was tapped. Consequently, most of the study trees at Chautauqua Bottoms tended to be multi-stemmed whereas trees at Thunderstorm Road tended to be single-stemmed. Tapping involved drilling a 9.5 mm diameter hole on the north-facing side of trees 1.0 m above the ground in the largest stem of each chosen tree. A 1.5 ml conical centrifuge tube (Fisher Scientific, Pittsburg, PA) with the bottom cut out was used as a spile and driven into the hole with a rubber mallet. At Thunderstorm Road, total sap volume (ml) was collected by routing tubing from the spile to a sample container at the base of each tree. At Chautauqua Bottoms a 5 cm long piece of tubing was placed in the spile to allow for placement of an SSC sample tube.

At both sites, when climatic conditions were favorable for sap flow (below freezing at night and above freezing during daylight) and the majority of tapped trees exhibited flow, fresh SSC samples were collected by placing a prelabeled 1.5 ml centrifuge sample tube on the end of the tubing. Sap samples were kept cold and returned to the lab for determination of SSC using a temperature compensated refractometer within 24 hours. A refractometer measures the total amount of solids in a solution, which has been shown to be highly correlated ($r = 0.99$) to SSC (Gregory and Hawley 1983).

Collections were made on four separate dates between 2001 January 22 and 2001 February 11. Sap volume (ml) was totaled for each tree over this same time period. Additionally, samples were collected over the same dates from sugar maple trees in a cooperators' sugaring operation located within 35 km from the silver maple plantings. In early spring of 2001, diameter (to the nearest 0.1 cm) at breast height (1.4 m above ground) of each stem of the silver maple study trees (except for those in a single block at the upland site) in the plantations was measured for use in correlation analyses against sap parameters. Sap sugar concentration was averaged for each tree over the four sample dates. Basal area (cm^2) was determined for each tapped stem. Total basal area for multi-stemmed trees was computed by summing the basal area of each stem.

Basal area and SSC of the trees were tested for significant sources of variation between sites, among provenances, among clones, and the interaction between site and provenance, and site and clone using the General Linear Model procedure of SAS (Statistical Analysis System Institute Inc., Cary, NC) at the alpha = 0.05 level. Total sap volume was tested among provenances and clones for trees at Thunderstorm Road. Type III error was used to calculate mean squares because it more conservatively

partitions variance especially for design imbalances related to missing data. Because of mortality and the lack of sap production of trees in some clones, only 49 clones were used in the analysis. Means are reported using LSMEANS that adjusts for missing plot data. Sap sugar concentration and volume were tested for significant correlations to diameter, basal area, and latitude and longitude of geographic origin. Sugar maples were tested for differences in SSC among individual trees by using each of the four sap collection dates as replicates and performing an analysis of variance.

RESULTS

Out of 520 plots originally planted at both sites, 447 plots had living trees. Some clones exhibited poor survival at both sites. Those clones tended to be members of provenance 13 arising from central Ontario. At the upland site all remaining clones were represented. However, at Chautauqua Bottoms two clones from central Kansas failed to survive.

Silver maple SSC did not differ among sites ($p = 0.134$) and averaged 1.51 percent (1.53 percent at Thunderstorm Road and 1.49 percent at Chautauqua Bottoms). There were highly significant differences ($p < 0.0001$) in SSC among provenances (ranging from 1.36 percent to 1.63 percent) and among clones (ranging from 1.23 to 1.87 percent). Significant interactions between site*provenance ($p = 0.041$) and site*clone ($p = 0.010$) existed in SSC indicating variable performance and rankings of clones and provenances depending on site (table 1).

Considering individual silver maple trees, SSC ranged from 0.87 to 3.18 percent at Thunderstorm Road and from 1.00 to 2.08 percent at Chautauqua Bottoms. In comparison, there were significant differences among individual sugar maple trees ($p < 0.001$) in SSC with an average of 1.87 percent and a range from 1.13 to 3.18 percent.

Basal area (cm^2) per tree did not differ among sites ($p = 0.093$) and averaged 65.7 cm^2 per tree. There were highly significant differences ($p < 0.001$) in basal area among provenances (ranging from 34.1 cm^2 to 90.6 cm^2) and clones (from 20.3 cm^2 to 114.8 cm^2). Significant site x provenance ($p = 0.195$) and site x clone ($p = 0.343$) interactions did not exist for basal area.

Sap volume differed ($p < 0.001$) among provenances and among clones (table 1). Sap volume ranged from 0.0 to 9048 ml among clones.

When considering both sites, significant positive correlations existed between sap parameters and tapped stem diameter, tapped stem basal area, and tree basal area (table 2). SSC tended to increase with increasing longitude of the trees origin but the correlation was relatively weak (0.122). SSC was not significantly correlated with latitude. Sap volume tended to increase with decreasing latitude and increasing longitude or origin.

DISCUSSION

There was considerable variation among provenances, clones, and individual silver maple

Table 1.—*Provenance means and rankings of sap sugar concentration (SSC) and sap volume (ml) by site*

	CHAUTAUQUA BOTTOMS		THUNDERSTORM ROAD			
	SSC		SSC		Volume	
Provenance	Percent	Rank	Percent	Rank	(ml)	Rank
08 (s. central WI)	1.61	1	1.63	3	1,433	4
01 (s. central MS)	1.58	2	1.67	2	3,755	1
11 (northeast PA)	1.55	3	1.52	5	1,988	2
05 (s. central IN)	1.53	4	1.49	6	1,616	3
12 (s. central Ont.)	1.52	5	1.73	1	1,130	6
07 (s. central WV)	1.48	6	1.39	12	156	12
17 (s. central NY)	1.47	7	1.31	13	1,067	7
06 (central IA)	1.47	8	1.63	4	568	9
15 (central NH)	1.45	9	1.48	7	307	11
16 (northwest VT)	1.41	10	1.42	10	386	10
18 (southwest VA)	1.37	11	1.45	9	629	8
20 (central KS)	1.35	12	1.45	8	1,161	5
13 (central Ont.)	1.29	13	1.42	11	0.4	13
std. error	0.04		0.06		262	

trees in SSC. High sap sugar concentration is an important factor in the production of maple syrup because the majority of annual production costs are attributed to energy required to process the sap to syrup (Huyler 1982). The sugar maple sampled in this study had an average SSC (1.87) and range (1.13 to 3.18 percent) that was comparable to previous studies (Gabriel 1972, Walters 1982, Heiligmann and Winch 1996). In comparison, average SSC for silver maple was somewhat reduced at 1.51 percent which was similar to the 1.43 percent SSC that was found within a different more diverse planting of silver maple (Keeley and others 2001).

For the current study, SSC ranged from 0.87 to 3.18 percent for individual trees suggesting that gains could be made by selection. Considering that these trees had been grown at relatively high density in these plantings that restricted full crown development, the full potential of sap performance may have been limited. Additionally, the interactions between site and provenance and site and clone suggest that care must be taken when making selections for high SSC because relative performance is dependent upon site conditions.

Sap volume also varied significantly with large differences among provenances and among clones. Average sap volume per tree was relatively small but trees of this small size would not normally be tapped in a commercial maple sugaring operation. Sap volume tends to increase with increasing tree size so it should be expected that increasing sap volume should be realized as a tree grows larger at least while a tree remains vigorous and does not become decadent.

With large sugar maple trees the amount of sap volume a tree produces tends to be relatively consistent from year to year (Morrow 1952, Marvin and others 1967). Sap volume was also positively correlated with SSC (r = 0.278). Marvin and others (1967), working with sugar maple, found a stronger relationship (r = 0.708) between sap volume and SSC over 18 years. Consequently, selecting for trees with either superior SSC, sap volume, or basal area characteristics should indirectly select for gains in the other characteristics.

Correlations between tree size and sap parameters were significant and positive suggesting that larger trees tended to produce greater volumes of sweeter sap. Blum (1971) reported similar findings when testing three sugarbushes where sap volume was correlated with basal area (r = 0.55 ranging to r = 0.62). However, SSC and dbh was found to be weakly correlated r = 0.13 to r = 0.19 (Larochelle and others 1998) or uncorrelated (Blum 1971).

In the current study, there was a stronger relationship between the basal area of the individually tapped stem and SSC (r = 0.419) or sap volume (r = 0.583) than basal area of the entire tree and SSC or sap volume (r = 0.185 and r = 0.468, respectively). This suggests that multi-stemmed trees impart more unaccounted variability in sap parameters. Although the individual stems within a multi-stemmed tree

Table 2.—*Pearson product moment correlations coefficients (r), significance level (p), and the number of observations (n) for sap sugar concentration (SSC), sap volume, diameter at breast height of tapped stem only (dbh tapped, cm), basal area tapped stem only (BA tapped cm²), basal area per tree (BA tree cm²), latitude (decimal degrees) and longitude (decimal degrees)*

	Sap volume	dbh tapped	BA tapped	BA tree	Latitude	Longitude
SSC						
r=	0.278	0.396	0.419	0.185	-0.023	0.122
p=	0.001	0.001	0.001	0.001	0.622	0.010
n=	230	401	401	401	447	447
Sap volume						
r=		0.541	0.583	0.468	-0.272	0.136
p=		0.001	0.001	0.001	0.001	0.036
n=		401	401	401	447	447
Basal area per tree						
r=					-0.191	0.088
p=					0.001	0.074
n=					413	413

are indeed attached, they appear to perform somewhat independently in sap parameters.

Examining provenance rankings at the two contrasting sites (table 1) shows some provenances are relatively consistent while others vary considerably. The same is true among clones. Some clones of provenance 13 from central Ontario failed to survive at both sites perhaps suggesting that they may not be fit to the relatively warm southern Illinois climate. The five provenances listed at the bottom of table 1 were among the poorest performers when considering SSC and volume together. These sources tended to originate from the furthest reaches from the planting site and on the fringes of the range of silver maple.

In some cases, there were relationships, albeit relatively weak ones, between the latitude and longitude of origin and sap parameters and basal area further suggesting that regional climate may not be suitable for all provenances or clones. Contrastingly, some clones appear to perform well at one site and not the other suggesting that while those clones may be fit to the regional climate, site-to-site variation within the region or even microsite variation must also be carefully considered when choosing stock.

CONCLUSION

Although silver maple has not traditionally been used for maple syrup production because of lower average sap sugar concentration, this study has demonstrated that the species possesses considerable genetic variability in sap parameters with individuals that are comparable to sugar maple. Silver maple grows rapidly on a variety of sites and can be readily propagated through vegetative means. Consequently, silver maple offers great promise as a species that could be integrated into sustainable forest product systems on marginal agricultural or in riparian filters to provide tangible ecological and economic benefits to landowners.

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PERFORMANCE OF BLACK WALNUT FROM THE CENTRAL HARDWOOD REGION IN CHINA'S YELLOW RIVER WATERSHED

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The introduction and evaluation of black walnut (*Juglans nigra* L.) into the People's Republic of China is a relatively recent development (Xi and others 1999). Small isolated black walnut plantings established by missionaries from the United States can be found throughout much of China; however, few written records document the growth of the surviving black walnut trees. In the early 1980s, the Chinese Academy of Forestry in Beijing and the University of Nebraska initiated a cooperative venture with the Friendship Nursery in Henan Province to exchange walnut and pecan germplasm. Wang (1996) estimates this venture resulted in more than 40 acres of black walnut plantings as nine demonstration plantings, nut cultivar trials, or scionbank orchards.

In March 1995, the Forestry Research Institute of China, the University of Nebraska, the Northern Nut Growers Association, and the Forestry Bureau of Luoning County called together nursery specialists and research horticulturists from China and the United States to evaluate the suitability of introducing black walnut and its economic potential for plantation culture in the semi-arid regions of China. Evaluation of existing plantings indicated black walnut is well adapted to the lower and middle regions of the Yellow River watershed.

Wang (1996) report expected annual growth of 4 to 6 feet for height and 0.6 to 1.0 inches in trunk diameter. He also reported walnut to be deep rooted with taproot length exceeding height, to be drought tolerant, and the only species with continual height growth during a 300-day drought. Because of the rapid growth and high value for the wood, the economic value of black walnut was estimated at 10-fold that for Chinese poplar (*Populus* spp.) and paulownia (*Paulownia tomentosa* (Thunb.) Sieb. & Zucc.). One conference recommendation suggested that black walnut should gradually replace both poplar and paulownia when reforesting farming areas in the Yellow River watershed. Under the Natural Forest Conservation Program initiated in 1998, China plans to convert 15 million acres of farmland to plantation forests within 10 years for soil erosion control in the Yangtze and Yellow River watersheds (Zhang and others 2000). Xi (1999) recently published guidelines in Chinese for the introduction, establishment, and management of black walnut.

In the fall of 1996, a 10-person delegation from the Chinese Academy of Forestry and the Henan Bureau of Forestry came to the Central Hardwood region to collect nuts from timber-type black walnut trees. Their objective was to establish multiple progeny tests and possibly identify the most appropriate areas in the United States for future seed purchases. Nuts were collected from plus-trees growing at the following five locations: the Pleasant Valley seed orchard (PVSO) located in Alexander County, IL and maintained by the Shawnee National Forest; the Tree Improvement Center arboretum (TICA) located in Jackson County, IL and maintained by Southern Illinois University; a grafted orchard (HGO) maintained by Leander Hay and located in Saline County, MO; Tuttle Creek Experimental Area provenance test (KPT) located in Riley County, KS and maintained by the Kansas State University; and several natural stands located near the Blue Bird Nursery (BBN) in Colfax County, NE. When possible, the source of the maternal germplasm was determined from plantation records or cultivar histories to identify county and state of origin,

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plant hardiness zone (Cathey 1990), and normal annual total precipitation and mean length of frost-free period (Hicks 1998). Plant hardiness zones are 4a for BBN, 5b for KPT and HGO, 6a for TICA, and 6b for PVSO.

Nuts from 37 individual trees (half-sib families) were cleaned and floated at the University of Nebraska. Sound seed was flown to Beijing, stratified for 3 months, pre-germinated, and planted into containers in a greenhouse. In-leaf seedlings of most families were transported to sites in plant hardiness zone 6 (Beijing provenance), 7 (Shanxi provenance), and 8 (Henan provenance) (Widrlechner 1997).

The Beijing progeny test (40.5 N, 116.5 E) is located on a first terrace site near the Badaling station of the Great Wall. The site has loam soils and subject to periodic flooding. The Shanxi progeny test is part of the XiaoYi Nursery in the TainYuan Basin. The site has deep, slightly alkaline, dark brown soils and mean annual precipitation of 18 to 20 inches with a 160- to 180-day growing season. The Henan progeny test (34.8 N, 113.5 E) is part of the Friendship Nursery near Zhengzhou. Planting site has a deep sandy loam with a 60-foot deep water table, mean annual precipitation of 23 to 24 inches falling primarily in the fall, and a 220-day growing season.

In-leaf seedlings of 30 or more families were planted in June 1997 in each progeny test using a randomized block design with row plots within each of four blocks. Planting design used four-tree plots at Henan and Shanxi tests and five-tree plots at Beijing test. Tree spacing is nominally 10 feet apart within row and 12 to 15 feet between rows. Within row vegetation was controlled for 4 years by hand cultivation. Survival, stem height (H), and trunk diameter (D) have been determined annually. Stem volume was estimated as D^2H. For each planting site, the performance index of each family was calculated as the average of the individual family percentile rank for stem height, diameter, and volume. Percentile rank within plantings was calculated as ((family mean – mean of poorest family)/(mean of best family – mean of poorest family) for each variable.

After four growing seasons, survival of the walnut seedlings has exceeded 95 percent at both the Henan and Shanxi plantings. Survival among families at the Beijing progeny test range from 20 to 95 percent. Mortality is the result of a summer flood in 1997 when 26 percent of the seedlings died and then failure to overwinter when an additional 17 percent of the seedlings died. In the spring of 1999, the ambrosia beetle (*Xylosandrus germanus*) attacked many of the surviving saplings causing extensive stem dieback and resprouting but little additional mortality.

The Henan progeny test has had the most rapid tree growth with average tree heights ranging from 16 to 23 feet among families after four growing seasons (table 1). Because of the rapid height growth, the trees are exceedingly thin with average trunk diameters of only 1.1 to 1.5 inches. In the fall of 2000, many of the trees showed premature defoliation in response to walnut anthracnose (*Gnomonia leptostyla*). Although anthracnose has relatively little impact on growth of walnut in the Central Hardwood region (Van Sambeek 2003), the longer growing season in China will likely result in a higher incidence of premature defoliation and a greater impact on tree growth. In addition, future growth in the Henan progeny test is likely to be slowed following the removal of all lateral branches during August 2000 and used as scionwood in the nursery.

The Beijing progeny test had the most variable height growth ranging from 14 to 24 feet among families (table 1). Part of the variation is in response to the ambrosia beetle that caused extensive stem dieback and subsequent basal sprouting in the spring of 1999. Weber and McPherson (1984) indicate this insect was introduced from eastern Asia into the United States. One growing season after being attacked, most trees had produced basal sprouts that were nearly as tall as the pre-attack tree heights. Weber (1981) reported similar basal sprouting and height growth on walnut saplings attacked by the ambrosia beetle in the Central Hardwood region. The walnut saplings in the Beijing progeny test are also exceedingly thin with average trunk diameters of only 0.7 to 1.3 inches.

The Shanxi progeny test had the slowest height growth and tree height among families averaging between 11 and 17 feet (table 1). As found in the other two progeny tests, the trees in the Shanxi progeny test were also exceedingly thin for their height with average trunk diameters of 0.7 to 1.5 inches.

No families were consistently the best or poorest performers across all three plantings (table 1); however, computing a performance index from percentile ranks did allow us to rank

Table 1.—*Black walnut progeny collection number and name; site of nut collection; origin of maternal germplasm by state, plant hardiness zone, annual rainfall, and frost-free days; performance index (mean percentile rankings); and mean height of 4-year-old saplings at three locations in Yellow River watershed of central China*

Progeny		Harvest	Maternal germplasm variables					Walnut age 4 height		
No.	Name[a]	site	Origin	P.H. zone	Rain-fall	Frost-free	Perf. index	Henan	Shanxi	Beijing
					-in-	-days-	-%-	-ft-	-ft-	-ft-
16	NC-5869 AA-34	TICA	IL	6a	41	200	0.90	22.2	14.4	——
27	NC-6134 IB-07	TICA	TN	7a	52	180	0.80	20.4	16.6	——
20	NC-6616 B#1	PVSO	IL	6b	45	210	0.73	23.2	14.4	——
23	Native Hay	HGO	MO	5b	40	180	0.72	20.9	13.5	23.0
4	NC-6600 B#4	PVSO	TN	7a	50	200	0.71	21.7	——	23.3
29	NC-6603 B#3	PVSO	TN	7a	50	210	0.68	20.4	13.3	——
8	NC-6138 PB-09	TICA	IA	5a	32	170	0.67	22.1	16.3	17.4
13	Carb #1	BBN	NE	4b	30	160	0.65	19.2	16.2	21.0
15	Cheeks #2 B#3	HGO	unk	unk	unk	unk	0.61	19.5	13.3	24.3
21	NC-5872 CC-20	TICA	MO	6a	45	190	0.59	21.8	13.3	——
24	Big Tree Clark	BBN	NE	4b	30	160	0.56	21.9	12.9	17.7
9	Kwik Krop	HGO	KS	6a	40	200	0.54	19.9	13.0	18.4
33	NC-6609 B#1	PVSO	KY	6b	48	200	0.54	——	14.4	18.4
22	NC-6622 B#2	PVSO	TN	6b	50	190	0.52	19.4	14.0	——
3	NC-5869 AA-30	TICA	IL	6a	41	200	0.52	——	12.3	21.3
6	KS-2707 #1	KPT	MO	6a	44	200	0.51	21.2	13.4	20.3
12	NC-5875 EE-20	TICA	MI	5b	34	150	0.49	20.7	14.0	19.7
28	Walter Jackson	HGO	KS	5b	37	190	0.47	17.7	15.2	——
1	KS-2707 #3	KPT	MO	6a	44	200	0.46	17.4	14.2	20.3
36	Clarkson #3-2	BBN	NE	4b	30	160	0.45	22.7	——	19.0
11	Mixed seedlot	HGO	MO	5b	40	180	0.36	19.3	14.0	18.0
7	NC-6160 KK-13	TICA	IN	5a	38	170	0.36	20.5	13.3	14.4
18	NC-6164 FA-07	TICA	IN	5a	38	170	0.34	16.9	14.3	——
2	NC-6176 MB-09	TICA	VA	7a	40	180	0.33	20.6	11.2	17.2
10	NC-6163 OA-09	TICA	IN	5a	38	170	0.31	16.0	13.1	20.0
26	NC-6134 IB-05	TICA	TN	7a	52	180	0.25	17.7	14.0	——
25	Carb #2	BBN	NE	4b	30	160	0.24	——	12.7	15.4
17	NC-5875 EE-20	TICA	MI	5b	34	150	0.24	18.0	12.9	——
37	Clarkson #2	BBN	NE	4b	30	160	0.23	17.6	12.5	16.4
5	Clarkson #3-1	BBN	NE	4b	30	160	0.11	16.3	11.7	17.0

[a] Maternal tree accession information and location within Pleasant Valley seed orchard (PBSO), Tree Improvement Center arboretum (TICA), Hay grafted orchard (HGO), Kansas provenance test (KPT), and Bluebird Nursery (BBN).

the families across the three plantings. The performance index of 0.90 for progeny #16 indicates it was not always the top performer for height, diameter, and volume; however, it does suggest it was one of the best performers in the progeny tests that it was included in. Likewise, the performance index of 0.11 for progeny #5 indicates it performed poorly but was not consistently the poorest performer.

All five locations (PVSO, TICA, HGO, KPT, and BBN) from which nuts were collected produced families with performance indices above and below the mean. Except for the BBN, the paternal genes (pollen parent) may have originated from across the Central Hardwood region. We determined the original source of the maternal genes for most of the open-pollinated families. Presumably by analyzing climatic variables from which maternal genes originated and the family performance index, we could identify the most appropriate provenances within the Central Hardwood region for future seed collections.

Maternal genes originated from 10 states (table 1). There was a trend for families originating from Tennessee to have the highest performance index and families from Nebraska to have

the lowest performance index. In several states, maternal genes originated from several plant hardiness zones. The performance index for families originating from plant hardiness zones 6a and 6b indicate they had above average performance; however, plant hardiness zones 4b, 5a, 5b, and 7a also had a few families with above average performance indices (fig. 1).

There was a positive correlation between mean annual rainfall from the region where the maternal genes originated and the performance index (fig. 2). Seven of the families originating from areas in the United States with greater than 40 inches of annual rainfall had above average performance indices. This was unexpected considering precipitation averages less than 25 inches annually at all three sites in China.

There was also a good correlation between performance indices and the length of the growing season or number of frost-free days in the region from which the maternal genes originated (fig. 3). Nearly all families where maternal genes originated from regions with 190 or more frost-free days during the growing season had above average performance indices

In conclusion, we found families throughout the Central Hardwood region that performed well in the Yellow River watershed of China. We also found that seed collections from certain geographic regions were more likely to yield highly successful families. Preliminary guidelines suggest seed should originate from areas within plant hardiness zones 6a and 6b that have a mean annual rainfall in excess of 40 inches with a growing season of 190 or more frost-free days.

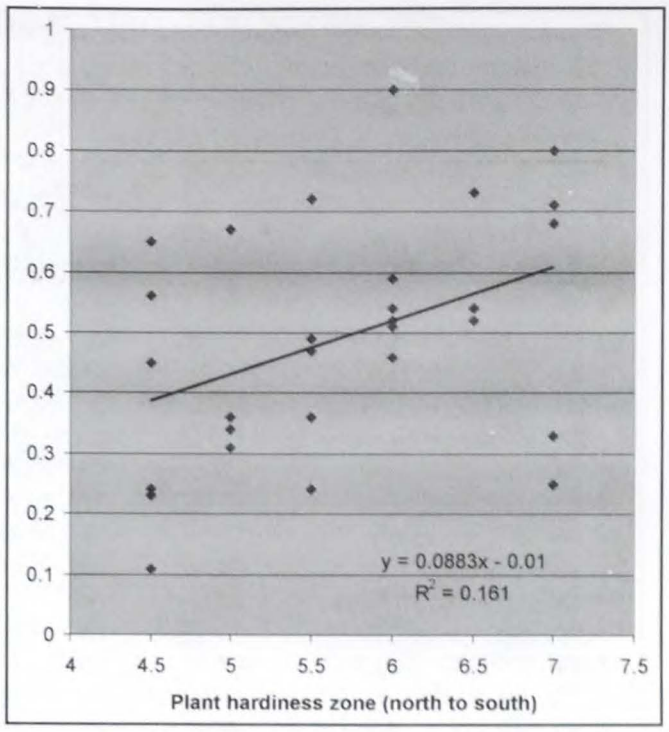

Figure 1.—*Performance index for open-pollinated families in Yellow River watershed and plant hardiness zone origin of maternal genes.*

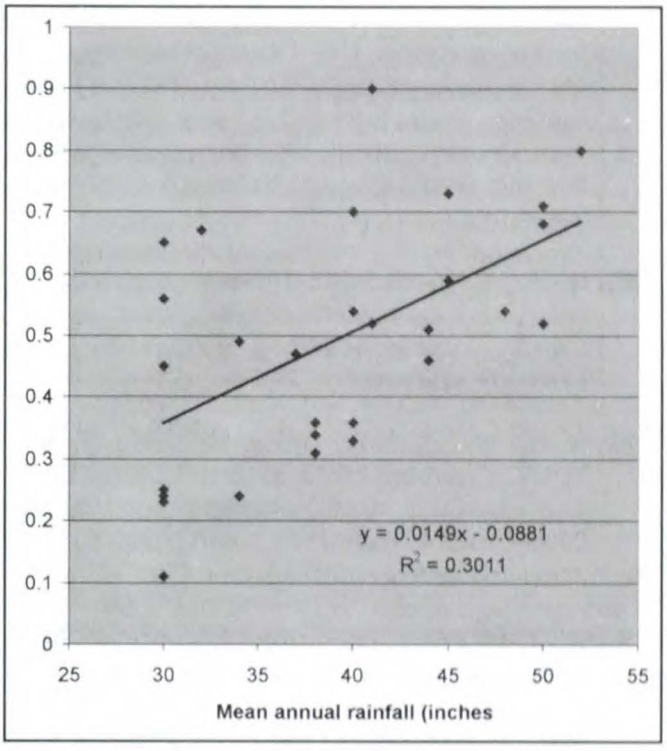

Figure 2.—*Performance index for open-pollinated families in Yellow River watershed and mean annual rainfall for region from which material genes originated.*

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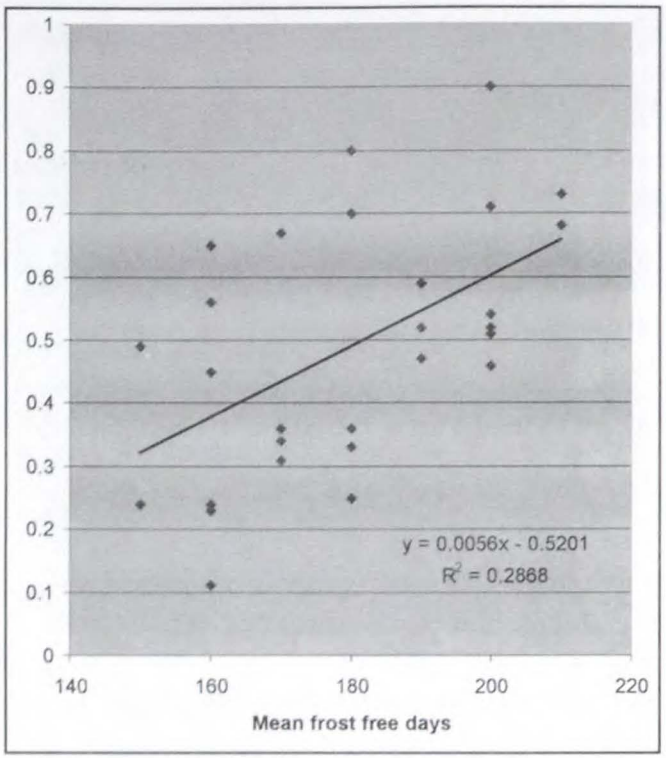

Figure 3.—*Performance index of open-pollinated families in Yellow River watershed and length of growing season where maternal genes originated.*

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THE GENETIC EFFECTS OF A DIAMETER LIMITED CUT ON BLACK WALNUT

Rodney L. Robichaud; Olin E. Rhodes, Jr.; and Keith Woeste[1]

Black walnut (*Juglans nigra* L.) trees are often selectively cut from forested stands based on their phenotype or size. This practice lowers the population density and possibly the genetic diversity of the species. Anecdotal evidence links this practice to an observed decline in the availability of high quality black walnuts.

To determine the genetic effects of a selective cut in a typical, even-aged, mixed hardwood stand, a population of 209 black walnut trees from Long Hollow, Hoosier National Forest, Crawford County, Indiana, was mapped, their diameter measured at dbh, their leaves sampled, and their DNA extracted. A random sample of 44 individuals was taken from this population and further subdivided into two size classes, those < 51 cm and those > 51 cm dbh. Twenty-six individuals represented the small class of black walnut, 18 individuals made up the large class. The different size classes were used to represent and analyze the effect of a virtual selective-cut where all the larger diameter class trees were removed.

DNA was isolated from these trees using an AutoGen™ NA-2000, automatic nucleic acid isolation system. A suite of 10 microsatellites with fluorescent labeled primers were used in genotyping these black walnuts (Woeste and others 2002). The PCR products from these 10 loci were run on Long Ranger® polyacrylamide gels using an ABI Prism™, 377 DNA Sequencer. Individuals were genotyped using Applied Biosystems GeneScan® v 3.1.2 with Genotyper® v 2.5. Once genotyped, these 44 selected trees were further analyzed using GDA v 1.0 (d16c) to calculate population and allelic statistics, levels of heterozygosity, mean number of alleles per locus, differences in F_{st}, and Nei (1978) distance. Chi-Square tests were used to test for independence and homogeneity of allelic distribution between the two size classes.

The large and small diameter classes of trees at Long Hollow were genetically similar. The number of alleles per locus ranged from 6 to 21 with a mean of 11.5 (table 1). Unique alleles were found within both the small and large classes of trees at all loci except WAG 79 (walnut-adenine-guanine dinucleotide microsatellite marker with 79 repeats). Two loci had unique alleles in the large size class only, while one locus had unique alleles in the small size class only. All the remaining loci had unique alleles in both classes.

A Chi-Square test of independence of the allelic distribution for each locus showed no significant difference between the two size classes. There was also no significant difference in the allelic distribution among loci for the two size classes based on the test of homogeneity of ratio (X^2_D=0.302, df=9, α=0.01). These results indicate that there is a high level of homogeneity for the frequency of alleles between the two size classes. Combined heterozygosity for the entire sample was 0.72 with an F_{st} of 0.001. The

Table 1.—*The number of alleles found within each microsatellite locus identified as number of adenine-guanine dinucleotide repeats*

Walnut A-G microsatellite locus	Number alleles
6	11
24	9
27	11
33	15
69	8
72	7
73	13
79	6
89	21
97	14

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observed heterozygosity by size class was H = 0.71 and H = 0.72 for the small and large size class, respectively. Nei (1973) distance measure for the two sub-samples was calculated to be 0.01. These results all indicate a high level of genetic homogeneity between the small and large size classes at Long Hollow.

The 44 individuals analyzed in this study were further subdivided into approximately equal sample sizes to evaluate the timing of pollen recruitment as the stand was maturing (table 2). We assumed the relationship between age and diameter was fairly constant within the stand. The results indicated a relative constant flow of new alleles into this population over time. Eight alleles from four loci were found to be unique to the Long Hollow stand, as compared to a previously analyzed provenance test of over 280 black walnuts, taken from 10 different states.

The black walnut population at Long Hollow was genetically diverse and homogenous in its allelic distribution. While there was no change in the level of heterozygosity when the large size class trees were (virtually) removed, a few unique alleles were lost. What effect, if any, the loss of these alleles would have on the remaining population, is unknown at this time. The rate of recruitment of new alleles into the population over time was found to be relatively constant. The impact of recruitment on stand diversity and regeneration are still under investigation.

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Table 2.—*Number of unique alleles by size class*

Subpopulations	Sample size	Dbh (cm)	Unique alleles[1]
LH1	9	7.1 – 27.9	6
LH2	13	28.0 – 46.0	4
LH3	11	46.1 – 55.0	11
LH4	11	55.1 – 71.0	7

[1]Unique alleles are unique to that sample in comparison to the others.

HEARTWOOD FORMATION IN FOUR BLACK WALNUT PLANTATIONS

Keith Woeste and Brian Beheler[1]

The amount of heartwood in black walnut (*Juglans nigra* L.) logs can vary widely, even among trees of the same age growing at the same location. There is little published data on the genetics, physiology, and development of heartwood in hardwoods, even though the volume of heartwood in a log can significantly influence its value.

We analyzed trees from four black walnut populations for growth, the volume of sapwood and heartwood, area of heartwood and percent heartwood area (table 1). Phenotypic correlations among the traits were determined to evaluate the feasibility of using such correlations for the improvement of black walnut breeding stock for percent heartwood area.

Tree diameter was measured at 1.4 m above ground (DBH) and/or at the stump after felling. The area and percent area of heartwood were calculated using increment cores (at DBH) or by measuring the heartwood and sapwood of felled trees at the stump and/or at DBH. The trees measured at Martell and Salamonie were the fastest growing trees in the stand (the trees remaining after silvicultural thinning). The trees measured at SEPAC and ShoNeff were the individuals being removed as part of a planned thinning.

The site conditions at all four plantations were excellent, as was reflected in the growth rate of the trees. There was no clear relationship between growth rate and percent heartwood. The depth of the bark averaged 0.51 ± 0.1 inches. The heartwood core in walnut was typically conical and significantly larger at the stump than at DBH. The heartwood cone was not centered in the tree but was shifted slightly to the north and east, and trees at the southern and western edges of plantations had slightly more heartwood than trees in the interior.

Growth (diameter and height) was consistently highly correlated with heartwood area (table 2). There was also a significant and strong positive correlation between heartwood area and percent area of heartwood (PAOH) both at the stump and at DBH. Estimates of the correlation between percent heartwood area and diameter were extremely variable, ranging from –0.45 to 0.74, and were frequently non-significant. Estimates of PAOH were probably most accurate when based on trees cut at DBH, because trees are rounder and the heartwood is more circular at this point than at the stump.

As a tree increases in diameter, the percent area of a cross section that was heartwood (PAOH) increased more quickly than the percent that

Table 1.—*Characteristics of the populations used in this study*

Population	Population type	Location	n	Year established	Growth (in/yr)	Mean % heartwood (± SD)
Martell	Grafted	Tippecanoe, IN	77	1975	0.3	46 ± 9
SEPAC	Seedlings	Butlerville, IN	76	1981	0.4	31 ± 11
ShoNeff	Seedlings	Stockton, MO	296	1974	0.4	42 ± 12
Salamonie	Seedlings	Lagro, IN	80	1963	0.26	47 ± 7

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was sapwood. The timing and rate of the expansion of heartwood and sapwood may depend on tree age, site, and genotype.

It was often difficult to identify a single point at which the sapwood ended and heartwood began. This was true both when samples were taken by increment core and when heartwood was measured directly from a log or stump. This difficulty in measurement complicated the estimation of PAOH. As a consequence, our estimates of PAOH varied considerably within a single tree, depending on how and where measurements were made.

The coefficient of variability (CV) for sapwood depth for even-aged trees was twice the CV for diameter. It is not clear how much of this variance was due to measurement error, how much was environmental, and how much was related to differences among genotypes. Precise measures of sapwood depth are important because small differences in sapwood depth created large differences in estimates of heartwood area and PAOH. Despite these difficulties, there were significant differences among families for heartwood area and PAOH in all of the populations we studied. However, without improved methods for measuring the amount of heartwood in a log and/or standing tree, genetic improvement for PAOH will be difficult, and these will be achieved only in the short term as a correlated response to selection for diameter growth.

Table 2.—*Spearman correlation coefficients for the relationships among growth and heartwood characters in four populations of walnut*

	Heartwood area (cm^2) at DBH	Percent area heartwood at DBH	Percent area heartwood at stump	Percent area heartwood (incr. core)
DBH	0.67-A[1,2] 0.76-C 0.94-D	NS-A NS-C	NS-A -0.23-C	NS-C 0.21-D
Total area	0.75-C 0.95-D	NS-C	-0.23-C	NS-C 0.21-D
Height	0.61-A 0.31-C	NS-C	0.31-A	0.21-A
Heartwood area	1.0	0.76-A 0.49-C	NS-C	0.37-C 0.47-D
Stump diameter	0.71-A 0.79-B 0.65-C	NS-C	0.23-A 0.74-B -0.45-C	NS-C
Heartwood area (at stump)	0.79-B 0.62-C	0.37-C	0.77-B 0.61-C	0.41-C

[1] All correlations are significant at $\alpha = 0.05$, NS = not significant.
[2] Letters after correlation statistic refer to populations from which estimates were obtained; A = SEPAC, B = ShoNeff, C = Martell, D = Salamonie, see table 1.

EMERGING HARDWOOD PEST PROBLEMS AND IMPLICATIONS FOR THE CENTRAL HARDWOOD REGION

Bruce D. Moltzan[1]

ABSTRACT.—Over the last 100 years of human action there has been a correlated increase in the movement of insects and disease to the new world. Chestnut Blight, Dutch Elm Disease, Gypsy Moth, Butternut Canker, and White Pine Blister Rust are stern reminders of historically devastating exotic invasions. Sudden Oak Death, Asian Long-horned Beetle, Oak Wilt, Red Oak Borer, and Oak Decline should be considered credible threats to the Central Hardwood region. The potential danger posed by these pests will necessitate increased plant inspection and regulation to reduce the impact of their arrival, establishment, and spread. Advanced risk assessment and detection linked with sound forest practice to promote species diversity will reduce the impact of unwanted exotic introductions.

The introduction of exotic invasive species to North America has often resulted in large scale tree mortality and the replacement of once dominant tree species. As compared to pests introduced to agronomic crops, the damage on forest ecosystems is greater because these tend to have long lived impacts (Liebhold and others 1995). Over the last 100 years the movement of people and increased trade in emerging global markets has led to a growing number of introduced biological pests (Pimentel 1986).

Co-evolution maintained overtime for a specific global location acts to sustain a dynamic state of equilibrium between populations of indigenous trees and their native biological pests. In other words, native tree/pathogen systems rarely produce a resistance that overwhelms the pathogen or a virulence that reaches epidemic proportions within a given host population. Barriers created by continental drift, oceans, mountains, and climatic extremes, to a large extent, maintain unequal distribution of most tree problems worldwide. When human movement overcomes these natural barriers, the likelihood of a non-native pest or host being deliberately or accidentally introduced will increase.

Fortunately, even if an invasive species does arrive in a non-native area the actual rate of establishment is very low. This can be attributed to the lack of reproducing populations or the absence of essential climatic conditions such as adequate moisture, humidity, and temperature preventing the development of persistent populations. Since both insect and microbial kingdoms have prolific adaptive capability, it should be apparent that the invasive agent if established would have an advantage over tree populations that lack co-evolved resistance.

Chestnut Blight, Dutch Elm Disease, Gypsy Moth, Butternut Canker, and White Pine Blister Rust all serve as significant reminders of exotic/introduced pests that successfully established in North America leading to the demise of once dominant tree species. The purpose of this paper is to discuss emerging and ongoing hardwood pest problems and their potential impact as they relate to the Central Hardwood region of the United States.

SUDDEN OAK DEATH

A disease now known as Sudden Oak Death (SOD) was first reported in 1995 in the central coastal region of California. The name of this disease was given in response to the rapid death occurring on tens of thousands of tanoaks (*Lithocarpus densiflorus* (Hook. & Arn.) Rehd.), coast live oaks (*Quercus agrifoli* Nee), and California black oaks (*Quercus kelloggü* Newb.) killed by a newly described fungus *Phytophthora ramorum* (Werres and others 2001). On these hosts, the fungus causes a

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bleeding canker on the stem. This is the most consistent and diagnostic symptom of the disease on larger trees and often develops before any visible foliage symptoms are evident. The distinction from other Phytophthora's is that SOD infections occur above ground up to 20 m (Garbelotto and others 2001, Rizzo and others 2002).

As of January 2002, SOD was known to occur only in California and southwestern Oregon, however, many new reports suggest it may have a broader distribution even into the mountain interior region of California. The geographic origin of this agent is currently unknown, but is believed to have been introduced from Germany and the Netherlands where it has only been reported on *Rhododendron* L. and *Viburnum* L. spp. (Werres and others 2001).

In ongoing tests, the fungus was shown to cause the disease on seedlings of northern red oak (*Quercus rubra* L.) and pin oak (*Q. palustris* Muenchh.) sent from the Central Hardwood region (Dave Rizzo, personal communication). This fact, coupled with widespread trading of rhododendron ornamentals, makes the introduction to the Central Hardwood forest a significant threat. Early detection will be important for successful eradication. Oaks defoliated early in the growing season by insects or pathogens may appear dead but leaves usually reflush later in the season.

Canker rots, slime flux, leaf scorch, root diseases, late season frost, herbicide injury, borer attacks, and other aliments may cause symptoms similar to those caused by *P. ramorum*. The most apparent of these are Oak Wilt, Oak Decline, and Red Oak Borer damage. In response to the SOD reports in southwestern Oregon, an aggressive campaign is presently under way to contain it by instituting a 100 m barrier zone followed by cutting and burning everything within the zone. If SOD were established in the Central Hardwood region, it most likely will be equally as aggressive. Often these approaches are not met with widespread acceptance by private landowners, so it may be a advisable to endeavor to educate the public of the potential threat.

Recently, the Animal and Plant Health Inspection Service, USDA, and others have issued quarantines restricting the movement of plants and other materials that might spread the SOD pathogen. This includes interstate shipment of potentially infested plants, foliage, wood, bark, and soil. Nursery inspections in the Pacific Northwest on rhododendron and azalea's now have more stringent requirements both in the nursery and prior to shipment. While this may be a good step forward, the fact remains that most states, particularly in the Central Hardwood region, cannot restrict the interstate movement of potentially infested material and many face budgetary deficits to adequately monitor for early detection.

Efforts are ongoing to determine whether climatic, topographic, and susceptible hardwoods similarly exist in the Central Hardwood region and whether these areas could be mapped in a risk assessment program. Currently, no infections on rhododendron sampled by the USDA Forest Service in the Southern Region, or samples sent to the Missouri Department of Agriculture have tested positive for this disease.

ASIAN LONGHORNED BEETLE

The Asian Longhorned Beetle (ALB), *Anoplophora glabripennis* (Motschulsky) is an exotic/invasive insect that represents a serious threat to the forest ecosystems of the Central Hardwood region and the entire United States. ALB is a native pest of China and Korea and has most likely been introduced into the United States within wood shipping crates packed from China. The ALB was first discovered in 1996 in New York City (Haack and others 1997) resulting in the removal of over 5,300 trees with recent infections (2002) reported near Central Park (Dennis Haugen, personal communication).

The ALB was also detected in appreciable populations within the greater Chicago area in 1998. Over 1,500 trees have been removed from infested urban areas with only a few new trees detected in recent surveys from infested areas. The beetle has also been detected in various ports and warehouses in other parts of the country and it is likely that other states have small isolated infestations that to date have not been detected.

The primary damage to the tree is caused by the ALB larvae that girdle the main stem and branches (Cavey and others 1998). If repeated attacks occur, the tree crown suffers dieback, eventually resulting in tree death. Damage from infestations in New York and Illinois has resulted in removal costs to State and Federal governments in excess of $80 million since the discovery of the infestations in 1996. If the ALB were to expand beyond the current quarantined areas of New York and Illinois, it has the

potential to damage trees nationwide, affecting such industries as lumber, maple syrup, nursery, commercial fruit, and tourism and accumulating over $650 billion in losses.

The ALB hosts are primarily maple species, but beetles have been recovered on horse chestnut, chinaberry, mulberry, poplar, cherry, pear, locust, willow, elm, birch, ash, and citrus (USDA 2001). The beetle is described as glossy jet black with a very smooth body and having up to 20 distinct white spots on its back. The ALB can range in size from 20 to 32 mm in length. The distinct bluish tinge on the tops of the beetles' feet is diagnostic. Adults are usually present May to October. The biggest problem with managing this pest is the lack of accurate detection by green industry workers. Often damage will occur high in the tree where many large exit holes may be overlooked.

Trade with China has increased exponentially over the past decade. In 2000, imported commodities from China to the United States exceeded $100 billion. As a result, the risk of this plant pest as well as the potential of other invasive insects, plant diseases, and weeds being introduced into the United States has increased as well. Presently, APHIS works with the Chinese authorities to take steps to prevent future infestations of the ALB and similar pests, including restrictions on softwood packing material from China and imposition of treatment requirements for these materials before they arrive in the United States.

To prevent further introductions, APHIS continues to analyze threats to our agriculture and develop rules for importing commodities based on the risks they present. These inspectors form the first line of defense against exotic plant and animal pests and diseases. All international passenger baggage, cargo, packages, mail, and conveyances are subject to inspection upon entry into the United States.

It remains to be seen whether native hardwood populations in the Central Hardwood region would be at risk. Because the majority of the beetle's life is spent deep within the heartwood of host trees, it is difficult to control using contact insecticides. Although costly and undesirable, the only assured method of eliminating the beetle is to cut and chip or burn infested trees and replace them with non-host species.

OAK WILT

Oak wilt is a vascular wilt disease that severely impacts oaks caused by an aggressive fungus *Ceratocystis fagacearum*. Some plant pathologists believe oak wilt to be exotic in origin arriving in North America in the early 1900s, however the fungus has never been reported from any country other than the United States making its status as an invasive unlikely. Presently, oak wilt is found throughout the Central Hardwood region and eastern United States. Although the disease is not found west of Texas, controlled studies have shown that most oaks in the red oak group and many western species are at risk should the fungus become established outside its normal range (Appel 1994).

Oak species killed by oak wilt include black oak (*Quercus velutina* Lam.), northern pin oak (*Q. ellipsoidalis* E.J. Hill), northern red oak, shingle oak (*Q. imbricaria* Michx.), and Texas live oak (*Q. virginiana* Mill.). The disease is less common on bur oak (*Q. macrocarpa* Michx.), white oak (*Q. alba* L.), and other oaks in the white oak group presumably due to the physiology and morphology of their sapwood.

The oak wilt fungus moves from tree to tree via two different pathways: underground root graft connections or overland spread by insect vectors. Initial and long-distance spread of the oak wilt fungus occurs when beetles carrying spores of the fungus come in contact with wound openings causing infection and death of the tree. The time period necessary for mortality to occur ranges from a couple of weeks after introduction in red oaks to several years in white oaks (O'Brien and others 2000).

Once established, interconnected roots or root grafts provide the means for the fungus to transmit infection from diseased to healthy trees. In the southern portion of the Central Hardwood region, it is likely that insect vectors play a more prominent role in transmission than root grafts. This is because there are many species of oak intermixed in the forest, and clay based soils may prevent widespread root grafting (Johann Bruhn, personal communication).

Symptoms include rapid leaf discoloration and wilting of the entire crown of the tree. Subtle off green color may be visible in the upper crown and is usually apparent throughout the summer. Bronzing from the distal portion of leaves is often delineated by a discrete margin between diseased and healthy leaf tissue. Leaves are cast rapidly as the infection progresses. If the

bark is stripped, it is possible to see brown streaking which is somewhat diagnostic.

Before costly controls are implemented, it is important to get accurate diagnosis from a qualified laboratory. Especially given there are many oak disorders that sometimes can be confused with oak wilt such as anthracnose, decline, and borer damage. Integrated management of oak wilt is built upon a variety of strategies to prevent new infection centers and limit the expansion of existing infection centers.

Once a center or pocket has been detected, removal of the dead trees will prevent a build up of inoculum in that area. Avoiding injury will also insure no openings for beetles to transmit the fungus. Recent studies suggest that as little as 15 minutes is all that is needed for fungal feeding nitidulids to find a wound. Proper timing of pruning should be restricted during peak periods of beetle activity and it should be noted that this may vary from north to south within the Central Hardwood region (Jennifer Juzwik, personal communication).

In the Lake States, the use of vibratory plows may sever root-grafts in sandy loamy soils. The blade should penetrate deep enough to disrupt grafted roots preferably to a depth of 5 feet. In the southern portion of the Central Hardwood region however, clay based, rock soils may make plowing difficult, at least a 3 foot depth is required to have an impact. In high value trees, it may be appropriate to inject trees with a propocanizole, however this seems to have a much better success rate in the white oak group and once administered must be repeated every 3 years.

More information is needed on how oak wilt differs in the north as compared to the south. In addition, update of the present distribution of oak wilt is of economic interest to those in the Central Hardwood region who seek markets in Europe where strict embargo of oak material exist. As the Central Hardwood forests become more fragmented, oak will continue to pose a high risk to urban and rural landowners.

OAK DECLINE/RED OAK BORER

Oak decline has worsened in the Ozarks recently due to a few years of drought and the advanced age of many oaks. While the complex is native to the Central Hardwood region, the result of this phenomenon is an ever-growing number of dead and dying oaks. Oak decline is caused by many environmental factors that together place stress on oak trees. Some of these factors have been present in southern Missouri forest stands for several decades.

The majority of trees affected by oak decline are in the red oak group. Within this group, the black oak and scarlet oak (*Quercus coccinea* Muenchh.) are the most severely affected. Many of these trees are over 70 to 80 years old and are growing in shallow, rocky soil on broad ridges or south and west-facing slopes (Starkey and others 1989). Often the stands are crowded with large numbers of mature trees. Trees growing under these conditions are competing heavily for limited water and nutrients. Over several years, a stressed tree becomes less vigorous, producing dwarfed, sparse foliage, and a thin crown. Branches in the upper crown begin to die back from the tips as tree growth is reduced. Various diseases and insects attack these weakened trees, stressing them even further.

In 1999, the USDA Forest Service estimated severe red oak decline and mortality existed on approximately 19,000 acres of the Ozark National Forest in northwest Arkansas. By June 2001, the estimated area of severe damage was approximately 300,000 acres. The numbers of red oak borer attacks had increased to unprecedented levels of 300 to 500 attacks per tree, compared to a more typical rate of less than 10 attacks per tree. The level of oak decline and mortality in Missouri was not as severe in 2001 as that observed in Arkansas, but there was increasing evidence of deteriorating conditions.

USDA Forest Service surveys in 2000 had estimated more than 16,000 acres on the Salem and Potosi Ranger Districts of the Mark Twain National Forest have high red oak mortality. *Armillaria* and red oak borers were clearly associated with dead and declining trees (Ross Melick, personal communication). By December 2001, the USDA Forest Service estimated that more than 100,000 acres on those two districts were seriously affected by decline and woodborer damage. Large acreages on other Ranger Districts were also seriously affected. Increasing levels of woodborer damage were observed in southern Missouri in 2001 on some private and state-owned land and on logs arriving at sawmills.

Tree crown dieback, reduced growth, and mortality caused by oak decline can have positive or negative impacts on the forest, depending on one's point of view. Cavities in dying trees can provide shelter or nesting sites

for various species of birds and other animals. Openings that are created in the forest canopy may allow changes to occur in the composition of tree species present. Forest stands that were predominantly oak may change to mixed pine-oak type, or even pine type, which may be desirable, given that pine once dominated many of these sites.

On the other hand, there may be serious negative impacts to wildlife due to reduced mast (acorn) production, reduced oak regeneration capacity, and fewer preferred species in the developing forest. In a recent study, mast yield was reduced by greater than 41 percent when compared with potential mast yield in stands without decline. An increased number of dead and dying trees adds to fuel loading in the forest and may lead to increased risk of wildfire. The economic impacts of the recent woodborer damage in Arkansas may force some smaller sawmills to close within the next few years.

Prior to 1880, the pine and pine-oak forests in Missouri covered about 6.6 million acres. Shortleaf pine (*Pinus echinata* Mill.) was often in pure stands or otherwise shared the landscape with black, white, scarlet, and post oaks. The rugged terrain discouraged railroads from laying track in the region. However, as demand for lumber continued to increase, and the forests of the eastern and Great Lakes States declined in lumber production, lumber companies and railroads turned their attention in the 1880s to southern Missouri and northern Arkansas.

By the 1920s, most of the large trees had been cut, and the lumber boom ended. Both the people and the land were heavily affected by changes at that time. Young oaks in the understory of the former pine forests grew rapidly and were an obstacle to grazing. Some farmers used intensive grazing by goats and sheep to cut the oak sprouts, but most burned the woods to keep the hardwood thickets down and encourage grass growth. Such burns exposed the thin Ozark soils to erosion that removed nutrients needed for plant growth. Pine regeneration was limited under these conditions. Logging had removed most of the mature pines that could provide seeds. Frequent burning killed pine seedlings, but encouraged hardwoods to re-sprout thereby facilitating the forest we now see today.

Although many diseases and insects are involved with oak decline, the most common disease agents are *Armillaria* and *Hypoxylon* fungi, and the most common insects include the red oak borer and two-lined chestnut borer. All of these organisms are native to Missouri. *Armillaria* is a fungus causing white rot of woody roots and is found wherever trees are growing. *Armillaria* typically acts as a decomposer, decaying coarse woody debris that has fallen to the forest floor.

When trees become stressed or wounded, *Armillaria* can act as an aggressive parasite attacking the tree's root system. Infected roots no longer effectively take up water, particularly in drought-stressed soils, resulting in a reduction of live tree crown. If drought conditions persist or worsen, *Hypoxylon* cankers begin to form on the main stems of trees. Ordinarily, tree bark defends against attack from external fungal invaders like *Hypoxylon*. However, when trees are stressed, they cannot effectively maintain their defense and infection occurs. Once the canker has girdled the stem, portions of the tree above the canker will die.

The red oak borer is a reddish brown beetle that is 20 to 32 mm long and has antennae as long as or longer than the rest of its body. It has a 2-year life cycle, and most adults of this beetle emerge from their host trees in June and July of odd-numbered years. Large numbers of these adult beetles are expected to emerge in 2003. After mating, the adult female deposits her eggs in crevices in the bark of host trees, usually red oaks. The eggs hatch and young larvae chew their way through the bark and into the tree, where they will spend the next 2 years.

Red oak borer larvae, which look like white wormlike grubs, create a wide cavity in the phloem layer just under the bark during the first summer and the following spring. By the middle of the second summer, larvae begin burrowing into the sapwood and often go deeper into the heartwood of the tree. Each larva creates a $^3/_8$-inch diameter tunnel that extends for several inches by first angling upward away from the bark and then turning straight up through the wood. The adult red oak borer emerges from the tree in the third summer by chewing open a $^1/_2$-inch diameter oval hole in the bark near where it entered the tree as a young larva.

Periods of reduced growth of oaks, due to drought in most cases, have occurred in the Ozarks in 1901, 1909-1911, 1923-1924, 1935-1937, 1953-1954, 1972, and early 1980s, an average of about once every 14 years. Mortality due to oak decline was common in southern

Missouri following droughts of the 1980s, although not at the scale of current conditions. Oak decline management will need to seek to increase diversity of tree species and take into account only those species that are appropriate for the conditions of the site where they will be growing. Most poorly-formed trees and trees with unhealthy crowns should be removed, but it is beneficial to keep some standing dead trees and trees with cavities or wildlife habitat. For example, cavity-nesting birds such as woodpeckers are important predators of wood-boring insects.

Landowners may avoid oak decline effects by regenerating oak stands (harvest mature trees and allow seedlings and sprouts to re-stock the site) before trees become less vigorous due to old age. White oaks and post oaks are slower-growing and longer-lived than black and scarlet oaks. Under very good growing conditions, white oaks may live up to 600 years, while black oaks may live up to 150 or 200 years. Their life spans under most Missouri conditions, however, are typically much shorter.

Some people have argued that, because oak decline is a "natural" phenomenon, we should let nature take its course. But current forest conditions are partly the result of past management practices that created "unnatural" forests that are not healthy enough to withstand disease and insect attacks. Active management plans that restore and maintain the appropriate species on appropriate sites will improve forest health and reduce the need for drastic management intervention in the future

GYPSY MOTH

Further on the horizon looms the gypsy moth (*Lymantria dispar* L.), an introduced insect that has caused extensive oak defoliation and mortality for many decades across the northeastern United States. The gypsy moth continues to expand its range and could arrive in Missouri sometime in the next 15 to 25 years. Like oak decline, the gypsy moth's primary targets are the oak species. Trees that are already stressed by drought and oak decline will be less able to withstand repeated defoliation by gypsy moth caterpillars. Managing forests in the presence of gypsy moths requires reducing the number of trees of susceptible species and thinning out the least vigorous trees. The actions we take now will not only affect how our forests respond to oak decline, but will affect how well these forests survive the gypsy moth when it reaches Missouri.

SUMMARY

There is a need for all plant regulating agencies throughout the world to continually assess the potential risk to exotic pests, many of which are yet to be determined. Quarantines and the agencies responsible for maintaining them will need to be continually re-evaluated with increased movement of plant material. More research on forest pests in their country of origin may provide additional insight on the biology and management of these threats. A pro-active approach now that seeks to anticipate new arrivals will favor early detection giving foresters a greater window of opportunity to eradicate and prevent further spread.

Further, planting exotic tree species may prove to be an undesirable practice because exotic plants are highly susceptible to exotic pests (Pimentel 1986). Past experience with exotic invasions should move our thinking toward reducing our dependence on imported raw or processed wood products (Liebhold and others 1995). Forestry practice should minimize the use of one or only a few tree species and increase diversity in the Central Hardwood region to meet the demand for fiber in the future.

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USE OF USDA FOREST INVENTORY AND ANALYSIS DATA TO ASSESS OAK TREE HEALTH IN MINNESOTA

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ABSTRACT.—As a precursor to a regional assessment for the Upper Midwest, three variables were examined as measures of oak health in Minnesota between 1974 and 1990 using USDA Forest Service Inventory and Analysis data. Mortality was < 1 percent in the 1974-1977 inventory, but > 6 percent in the 1986-1990 inventory based on numbers of dead oaks per total oaks on plots with ≥ 25 percent oak in three forest types. Over two-thirds of the oaks measured during "leaf-on" period in both cycles were in relatively good condition based on live crown ratio (value > 30 percent). Damage incidence observed during "leaf-on" varied by forest type. Incidence of damage-free trees decreased with increasing tree size class and with increasing position in the canopy. Incidence of "decline-associated" damage increased with increased size and increased position in the canopy.

Oak (*Quercus*) species occur on over 5 million hectares of land in the seven states of the Upper Midwest (Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, and Wisconsin), and the volume of oak growing stock covering all age classes on timberland in the region exceeds 503 million m^3 (Juzwik and Schmidt 2000). Oaks provide food and habitat for wildlife, landscape trees, and wood for lumber and veneer; thus, supporting the health and sustainability of the oak forest resource is important.

There have been several published reports of oak mortality within the North Central region since the 1960s (Millers and others 1989). Oak wilt, caused by the fungus *Ceratocystis fagacearum*, and oak decline, caused by a variety of biotic and abiotic factors, are diseases that cause mortality at the landscape level (Oak 2001). In a recent survey of state foresters, in five of the seven Upper Midwest states, either oak wilt or oak decline was considered the major disease of concern (Billings 2000). High levels of oak mortality are anticipated in areas of Missouri, given recent drought conditions and large populations of red oak borers (Gill and Wiggins 2001, Lawrence and Moltzan 2001). Although recently initiated efforts have begun monitoring and documenting specific information on oak tree health in the region (USDA Forest Service 2002, Witter and others 2000), information on historical trends in the condition of oak tree health across the region is lacking.

Regional Forest Inventory and Analysis (FIA) units within the USDA Forest Service have been collecting data from thousands of field plots in forested, non-urban areas since the 1930s (Miles and others 2001). Although sampling and collection efforts were primarily designed to extrapolate amounts and distribution of forest tree species across large landscapes, data on various types of damage to the measured trees were also recorded. In the late 1980s, Starkey and others (1992) used FIA data to identify and compare the numbers of plots with oak decline in bottomland versus upland sites in the southeastern USA. An oak decline atlas layer resulted from this effort. Further work explored the variables that were consistently related to high incidences of oak decline (Nebecker and others 1992, Oak and others 1996).

The overall goal of our research is to obtain a region-wide assessment of oak forest health in rural areas of seven states in the Upper Midwest, USA, using existing data sources.

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Results of such an assessment could provide a starting point from which to evaluate future changes and a framework to support effective management of our oak forests (Janz 1999, McWilliams and others 2002). USDA Forest Service FIA data offer the most complete and consistent data for the region, and the data were made available to us through a special agreement with the North Central Research Station FIA unit. Data for the two most recently completed inventory cycles for Minnesota were explored as a preliminary study in preparation for the regional evaluation.

The objectives of this study were to develop and evaluate modified FIA variables that would allow us to:

1) determine the distribution and levels of oak tree mortality over time,
2) estimate general tree crown conditions and changes based on recorded live crown ratio,
3) quantify decline-associated damage, and
4) explore relationships between these three variables and forest type, tree size class, and position of the tree in the canopy.

DATA ACQUISITION AND MANIPULATION

The two most recently completed periodic inventories for Minnesota for which electronic data are currently available are cycle 4, conducted between 1974 and 1977 (Jakes 1980) and cycle 5, conducted between 1986 and 1990 (Miles and others 1995). The data are stored in FIA databases organized for the Oracle Database Management System, and are accessible with the Structured Query Language (Miles and others 2001). The plot database provides plot and stand-level information such as the date of measurement, plot location, condition, and surroundings. The condition class database documents the overall status or condition of the trees as well as the plot area itself. Finally, the tree database contains the data on the condition of each individual tree at the time of measurement. Variables considered most relevant to the study objectives were selected from these databases. Data were extracted, using SQL Navigator (Quest Software Inc., Newport Beach, CA), and were exported to a spreadsheet program for manipulation and summarization.

Because a number of plots in cycle 5 were modeled, only data from plots that were field measured were used. Since a significant number of oak trees occur in several forest types, the incidence of oaks in all forest plots was first explored. Three forest types (oak-hickory, maple-beech-birch, aspen) were found to contain 90 percent of all the FIA inventoried oaks in the state (Kromroy, unpub. data). FIA plots within this subset of data were then examined and those with ≥ 25 percent oak stems per total number of tree stems on the plots were identified and selected for further analyses.

VARIABLE SELECTION AND MODIFICATION

FIA field crews used numerous variables to document tree mortality and assess tree condition, including health, in cycles 4 and 5 (USDA Forest Service 1989, USDA Forest Service 1999). For this study, three variables were selected: Mortality, Live Crown Ratio (LCR), and Damage. Definitions and descriptions of subcategories within each variable were carefully reviewed and served as the basis for elimination, combination or other modifications in the creation of the specific variables for this investigation.

Mortality was defined by FIA as whether a tree was alive or dead at the time of field measurement. Live trees included those that had a recorded history of a new live tree (a remeasured living tree) or a living tree that grew into the plot since the previous cycle. Subcategories of "dead" included trees determined to be a dead-salvable tree, dead standing, stump of a dead-salvable tree, or a tree that had grown into the plot since the previous cycle and had died.

Live Crown Ratio (LCR) is recorded by FIA field crews as an estimate of the percentage of total tree height that supported live green foliage for each tree measured in each cycle. Only data collected during the growing season (May to September) were extracted for use and classification for this study. The LCR values (1 to 9) were grouped into three new classes representing 1 to 30 percent, 31 to < 60 percent, and ≥ 60 percent LCR. In using LCR as a coarse measure of tree health for this study, values greater than 30 percent were considered an indicator of good tree health.

Damage was assessed by FIA field crews for each measured tree for the presence of damage due to any type of biotic or abiotic agent or environmental stress. If the tree appeared healthy, with no apparent damage, "0" was recorded; any other number indicated presence of some type of damage (USDA Forest Service 1989, USDA Forest Service 1999). In cycle 4, 22 damage codes were available for crew use, compared to 52 codes in cycle 5 (some were host-species specific). For this study, damage codes were utilized in two different ways. Incidence of any damage (presence/absence)

was considered first. Further exploration of this variable involved grouping assigned damage codes according to their known association with forest tree decline ("decline-associated") (as defined by Manion 1991), and specifically, biotic and abiotic agents associated with oak decline (Wargo 1983, Tainter and Baker 1996) versus "all other damage" (non-decline associated) (table 1). Data for live and dead trees measured during the growing season were used.

DATA ANALYSIS AND SUMMARIZATION

Data on the three measures of tree health (Mortality, LCR, Damage) were analyzed for total populations of trees measured in cycle 4 and in cycle 5. When appropriate, subsets of these data were created to include information on individual oaks measured in both cycle 4 and cycle 5 ("remeasured trees"). Data for each study variable were grouped in several ways (i.e., by forest type, tree size class, and position in the canopy) and summarized for the state. Tree size classes were based on tree diameter at breast height (dbh) and categorized as: seedling-sapling (< 13 cm dbh), poletimber (13 to 28 cm dbh), and sawtimber (> 28 cm dbh). Position in the canopy, or "crown class" (Miles and others 2001), was grouped as intermediate and overtopped, codominant, and dominant trees. Mortality was further summarized at the county level.

Graphical presentations of the data were visually analyzed for apparent differences and/or trends. Changes in condition were considered only for remeasured trees. Tracking of individual trees may be critical to interpreting plot level changes in decline (Oak and others 1990). As a result, data used to calculate change were from substantially fewer plots and trees than were available in each individual inventory.

RESULTS

Less than 1 percent of 11,517 oaks measured during cycle 4 (1977) in Minnesota were dead, while 6.5 percent of cycle 5 (1990) measured trees (9,876) were dead. Due to the extremely low level of mortality found in the earlier inventory, no further summarization or analyses were done with the cycle 4 data. For cycle 5, the distribution of oak mortality in oak-hickory, maple-beech-birch, and aspen forest plots with ≥ 25 percent oak (656 plots) was summarized on a county basis (fig. 1).

In four counties in southeastern Minnesota (Fillmore, Houston, Wabasha, and Winona), where the number of plots ranged from 21 to 44, between 5 and 10 percent oak mortality was observed. In 11 north central counties, where 16 to 51 plots per county were measured, mortality ranged from < 1 to 5 percent for Aitkin, Becker, Todd, and Wadena Counties, 5 to < 10 percent for Cass, Crow Wing, Hubbard, Morrison, and Ottertail Counties, and > 10 percent for Beltrami County. In 22 other counties, where ≥ 5 percent mortality was found, the numbers of measured plots in each were relatively low (1 to 15) and county estimated mortality values could be misleading.

Table 1.—*USDA Forest Service Forest Inventory and Analysis (FIA) damage codes recorded for oak trees in Minnesota's cycle 5 (1990) inventory and their use in the creation of three damage type classes*

FIA reported damage by:			
Category	**Code(s)**	**Number of trees**	**Damage type class[1]**
Healthy	0	1,329	None
Insects	100's	75	Decline-associated
Diseases	200's	269	Decline-associated
Weather	300's	203	Decline-associated
Selected unknowns	900's	366	Decline-associated
Sub-group total		913	
Animal	400's	13	All other
Fire	500	8	All other
Suppression	600	43	All other
Logging-related	800, 820	22	All other
Abnormal growth	905, 908	300	All other
Miscellaneous or unknown	700's, 900	124	All other
Sub-group total		510	
All damage combined		1,423	

[1] Variable created for this study.

Because the number of oaks measured differed by forest type in the cycle 5 inventory (78 percent of total in oak-hickory, 14 percent in aspen, 8 percent in maple-beech-birch) mortality was examined within forest type. Mortality averaged 6 percent on 446 oak-hickory and 125 aspen forest type plots, and 10 percent on 85 maple-beech-birch plots. Mortality differed by tree size class within each forest type (fig. 2). In the oak-hickory and aspen forest type plots mortality was highest (8 to 10 percent) for the poletimber class. Mortality was highest and similar (11.0, 10.7 percent) for both poletimber and sawtimber size oaks in the maple-beech-birch type. Data on tree position in the canopy were not recorded for dead-standing or dead-salvable trees.

Live Crown Ratio (LCR)

The proportion of oak trees with different classes of LCR were similar for the two inventory cycles based on the total number of live oaks measured between May and September in each inventory (3,775 trees on 334 plots, cycle 4; 2,963 trees on 203 plots, cycle 5). Specifically, the percentages of trees in each LCR class were: 33 and 29 for 1 to 30 percent LCR, 62 and 64 for 31 to 60 percent LCR, and 5 and 7 for > 60 percent LCR for cycle 4 and 5, respectively (data not shown). The patterns of frequencies of trees across the three LCR classes were similar for forest type within a cycle, and general trends for the forest types were similar for both cycles, (fig. 3). In both cycles and all three forest types, the LCR class with the majority (58 to 74 percent) of the oaks was 31 to 60 percent.

Furthermore, the majority of oaks within each of the three tree size classes (seedling-sapling, poletimber, sawtimber) fell within the 31 to 60 percent LCR class (data not shown). Differences were found, however, when the distribution of oaks by position in the canopy (intermediate or overtopped, codominant, dominant) was considered. The frequencies of trees within canopy position class were similar for both inventory cycles, but differences in patterns were found across canopy position within each cycle (fig. 4). Not surprisingly, the intermediate and overtopped trees had the highest percent of trees in the lowest LCR category (1 to 30 percent LCR) and the dominant oaks were most commonly represented in the higher crown ratio classes (i.e., > 30 percent LCR).

Change in LCR class for individual trees between the two inventory cycles was examined based on 866 oak trees in 193 plots that were measured in both cycles. Of this subset, 124 oaks (14.3 percent) were classified as dead standing or dead salvable, 70 of which had

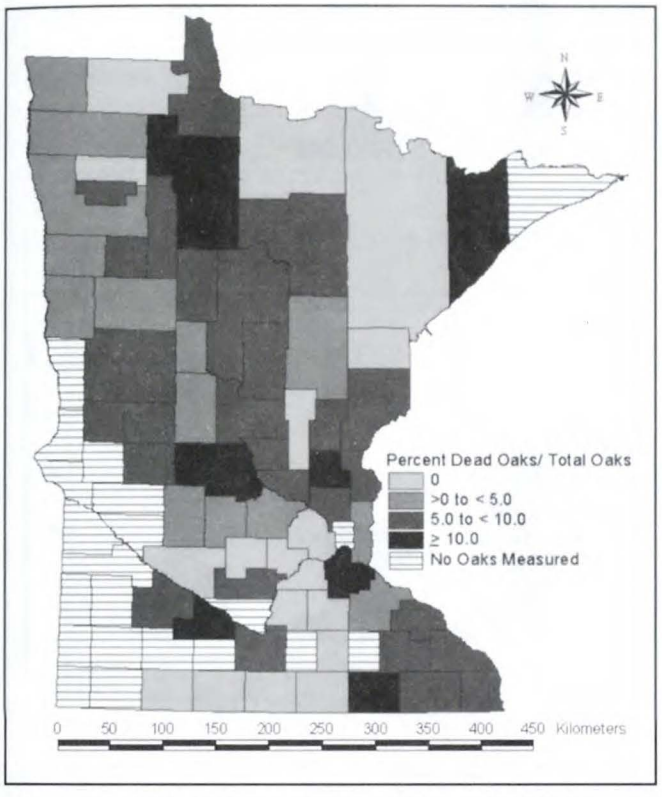

Figure 1.—*Percentages of dead oaks per total measured oaks per county in oak-hickory, maple-beech-birch, and aspen forest type plots with ≥ 25 percent oaks in cycle 5 inventory for Minnesota.*

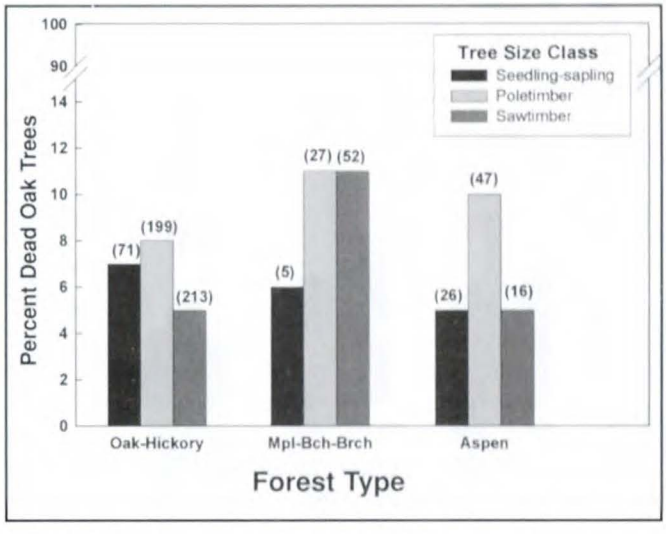

Figure 2.—*Percentages of dead oaks within three tree size classes in oak-hickory, maple-beech-birch, and aspen forest type plots for Minnesota cycle 5 inven-tory. Values in parentheses are actual numbers of dead oak trees.*

LCR's between 31 and 60 percent in the cycle 4 assessment. In addition, 111 oaks (12.8 percent) were recorded as stumps in cycle 5 (condition immediately prior to cutting unknown).

Incidence and degree of change in LCR was examined for the 631 trees that were recorded as live in both inventories (fig. 5). These changes were based on the LCR value recorded in the FIA database, not on the LCR categories that we created. No change in LCR was found for 38 percent of these trees between cycle 4 and cycle 5, while 43 percent of the trees had increases of 10 percent or more live crown, and 19 percent of the trees had decreases of 10 percent or more. For the live trees, differences were found by forest type, where 23 percent of oaks in aspen forests had some degree of loss of LCR between inventories, compared to 19 and 15 percent of oaks in maple-beech-birch and oak-hickory forests.

Conversely, over 50 percent of the oaks in oak-hickory forest type plots showed an increase in LCR. This compares to 37 and 40 percent of the oaks with a LCR increase in maple-beech-birch and aspen forest types. When considering changes in LCR by tree size class (for forest types combined), losses in LCR between the two cycles were similar (18 to 21 percent) across the classes, while increases in LCR were most common (49 percent) in the seedling-sapling size class and least common (38 percent) in the sawtimber trees. Finally, when considering changes in LCR by position of a tree in the canopy (for forest types combined) dominant

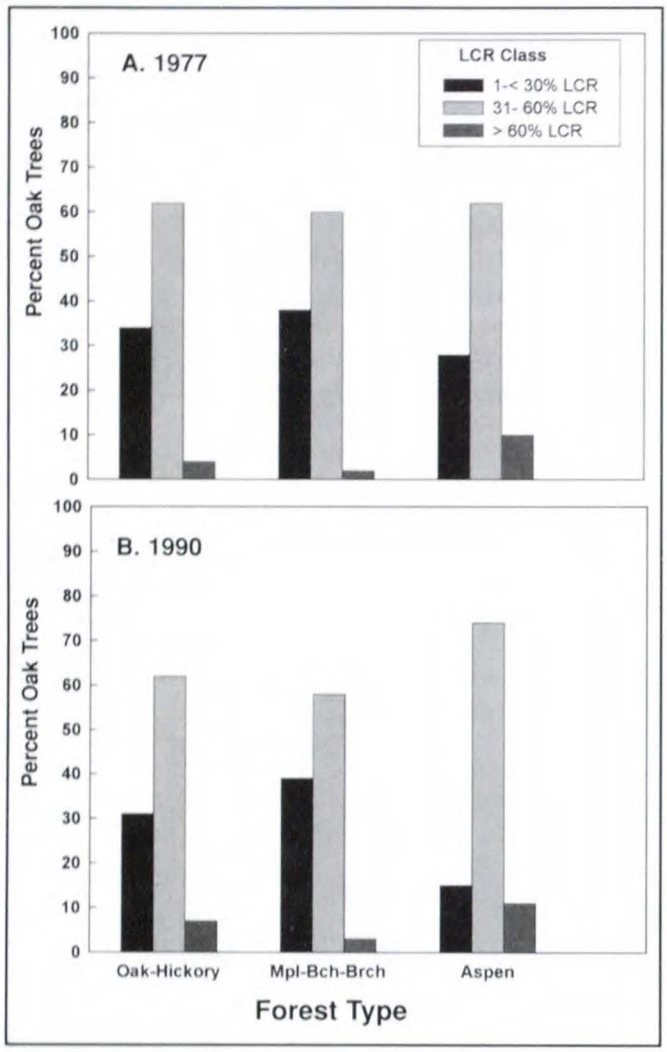

Figure 3.—*Percentages of oak trees among three live crown ratio classes (LCR), within oak-hickory, maple-beech-birch, and aspen forest types for two Minnesota forest inventories. A. 1977 = cycle 4, B. 1990 = cycle 5. Values are based on total number of oaks within each forest type.*

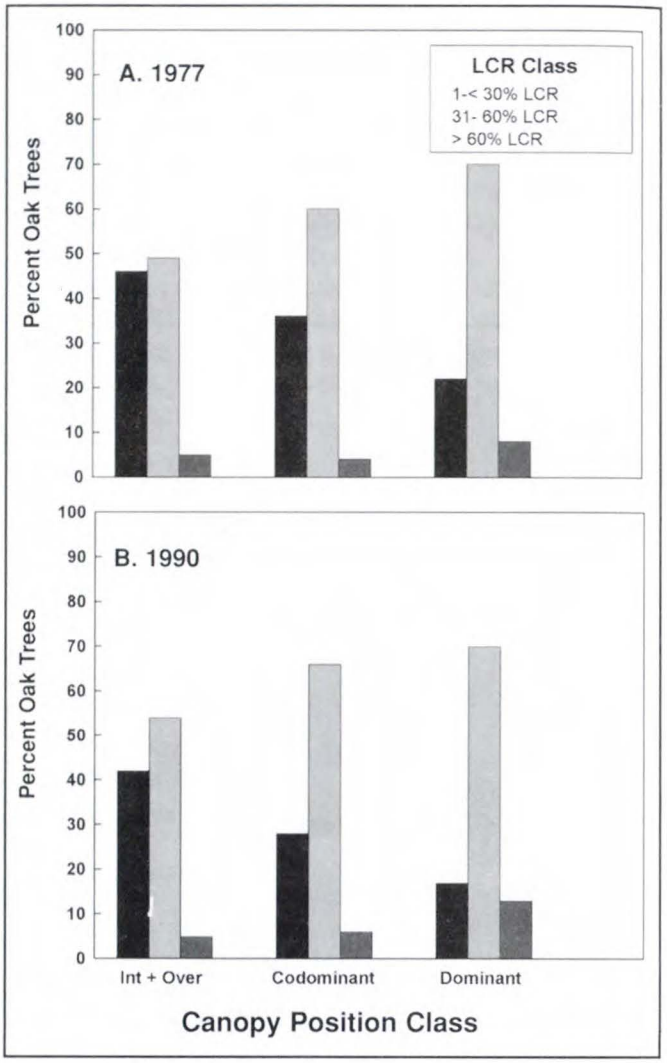

Figure 4.—*Percentages of oak trees among three live crown ratio classes (LCR), within intermediate and overtopped, codominant, and dominant positions for two Minnesota forest inventories. A. 1977 = cycle 4, B. 1990 = cycle 5. Values are based on total number of oaks within each forest type.*

trees had the highest occurrence of loss in live crown ratio, while intermediate and overtopped trees had the highest occurrence of increase in LCR values.

Damage and Decline

Because 85 percent of the oaks determined to have some type of damage in cycle 4 were assigned codes for which definitions could not be found, no further summarization or analyses involving cycle 4 damage data were done. In the cycle 5 databases, only a small percentage of trees with damage were assigned codes for which no definitions existed; such trees were omitted prior to data summarization.

Of 2,752 oak trees measured on 203 plots between May-September during cycle 5, 52 percent had some type of damage. Of the 52 possible codes, 32 were used to describe damage on the oak trees (only one used per tree); 10 of these 32 codes were used for less than five trees each. Damage incidence (presence/absence) varied by forest type with the highest incidence (55 percent) found in oak-hickory forest type plots compared to 42 and 43 percent in the maple-beech-birch and aspen plots, respectively (data not shown). Within each of the three forest types, "decline-associated" damage occurred more frequently than "all other" damage (fig. 6).

Differences in damage type occurrences were found among the three size classes within each forest type (fig. 7). The incidence of "no damage" decreased with the increase in tree size class (i.e., from seedling-sapling to poletimber to sawtimber). For example, incidence of "no damage" ranged from 90 percent in seedling-saplings to 42 percent in sawtimber size oaks in maple-birch-beech forest type plots.

Furthermore, the incidence of decline-associated damage increased with increasing tree size. For example, decline-associated damage ranged from 11 percent for seedling-saplings to 43 percent for sawtimber size oaks within oak-hickory forest plots. The incidence of "all other" damage did not show any trends with tree size, and levels were less than 25 percent within each size class and forest type.

Differences in damage type occurrence were also found among position in the canopy classes; the incidence of "no damage" decreased with increasing position of the trees in the canopy (fig. 8). For example, incidence of no damage ranged from 74 percent in intermediate and overtopped oaks combined, to 37 percent in dominant oaks on maple-beech-birch forest plots. Furthermore, the incidence of "decline-associated" damage increased with increasing position in the canopy. For example, "decline-associated" damage ranged from 19 percent in intermediate and overtopped oaks to 46 percent in dominant oaks within oak-hickory forest type plots. No positively- or negatively-correlated trends were evident for incidence of "all other" damage.

DISCUSSION AND SUMMARY

The utility and value of each of our variables in measuring oak health varied. Mortality, based on numbers of dead oaks per total oaks, was very low (< 1 percent) in the earlier inventory

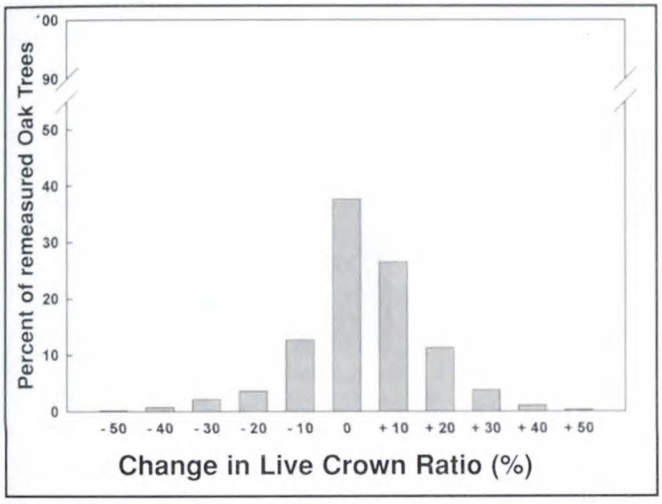

Figure 5.—*Frequency distribution by amount of change in live crown ratio (LCR) for live trees measured in May to September in Minnesota forest inventories cycle 4 and cycle 5 (n = 868).*

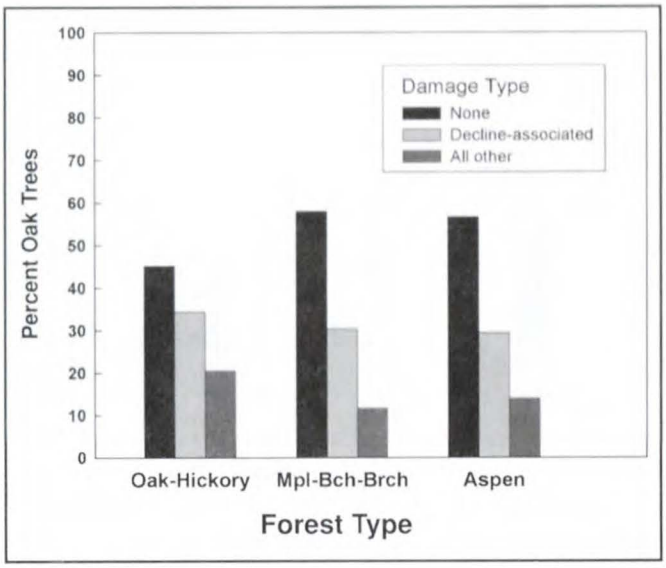

Figure 6.—*Percentages of oak trees within three damage classes.*

but apparently higher (> 6 percent) in the latter one and suggests that mortality increased over the time period considered. We consider mortality to be the most direct measure of oak forest health of the three variables used in this study. Mortality was a "straight-forward" calculation and required the least amount of interpretation of the FIA variables. In FIA published reports, the primary indicators of forest health are mortality estimates that are indexed as a percent of volume of the species (McWilliams and others 2002). The latter view of mortality gives a more direct estimate of economic impact, while our approach is more appropriate for oak health assessment.

In considering our results on the distribution of oak mortality across Minnesota it is important to keep in mind that the numbers of FIA plots and trees on which the mortality averages were calculated varied greatly. Our mortality estimates for the State may mask the impact of local mortality events such as those chronologically summarized by Millers and others (1989). However, it appears that greater than 5 percent mortality is common in the major oak growing areas of the State (fig. 1).

The damage variable provided a measure of the incidence of trees with damage as well as information on the type of damage, but only for the

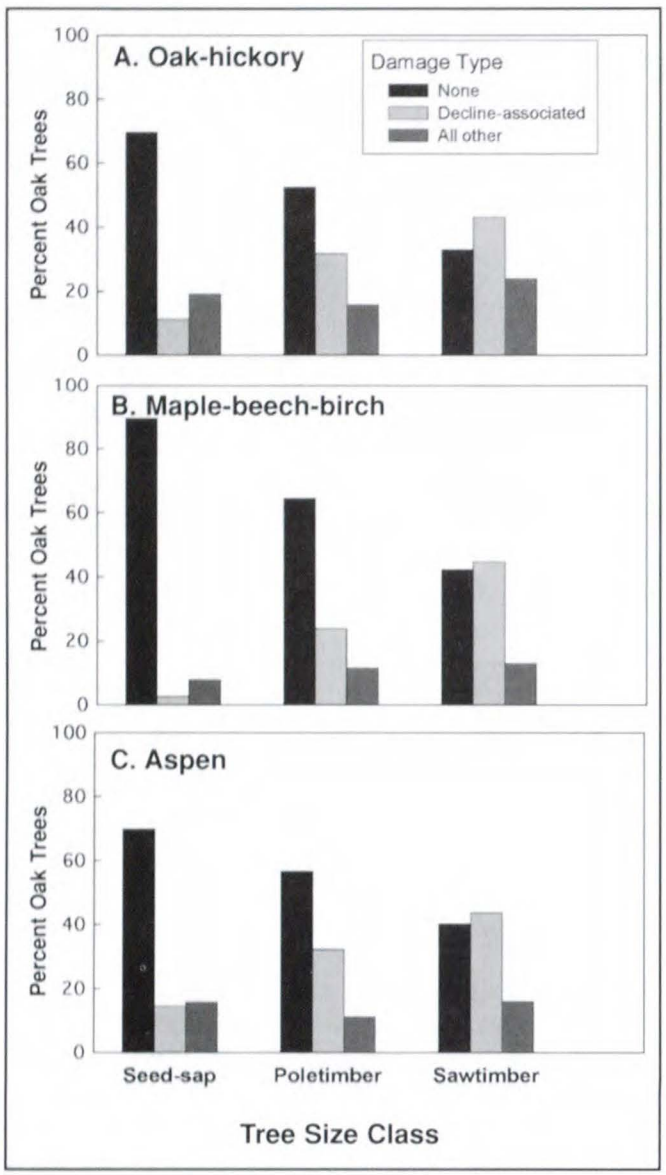

Figure 7.—*Percentages of oak trees among three damage classes within three tree size classes for Minnesota inventory cycle 5 for oak-hickory, maple-beech-birch, and aspen forest types.*

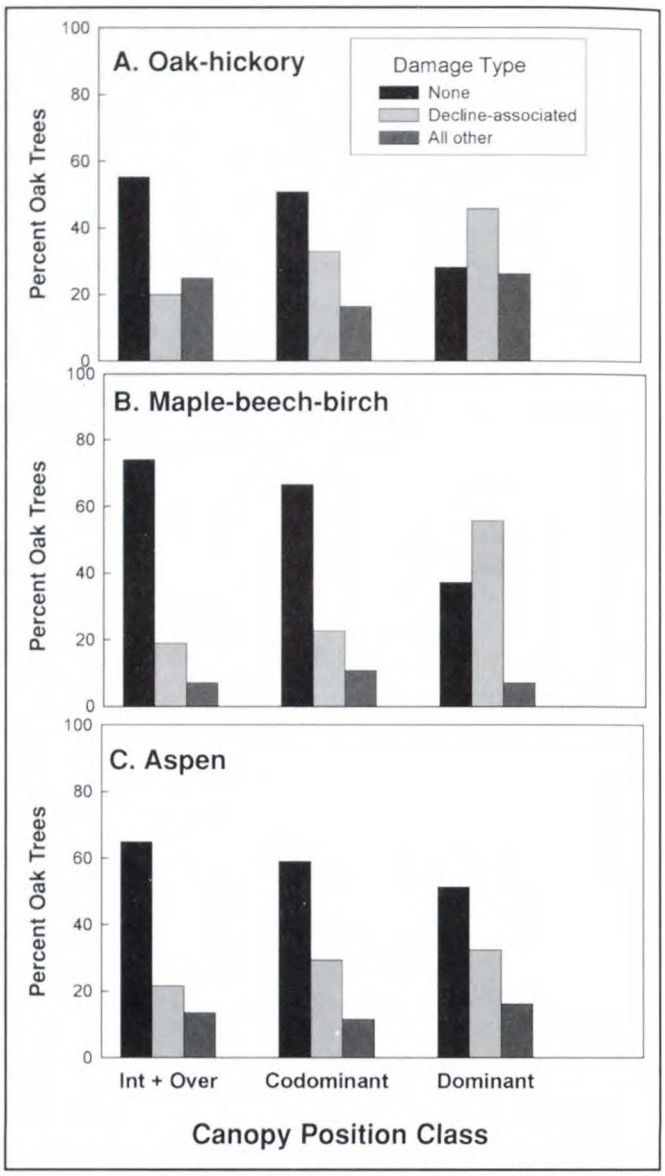

Figure 8.—*Percentages of oak trees among three damage classes within intermediate plus overtopped, codominant, and dominant canopy positions for Minnesota inventory cycle 5 for oak-hickory, maple-beech-birch, and aspen forest types.*

latter inventory due to prevalent errors in the earlier inventory. In cycle 5, over 50 percent of the oak trees had some type of damage, the majority of which (64 percent) are those commonly associated with oak decline. The incidence of trees with decline-associated damage increased both with increasing tree size and increasing position in the canopy. These observations generally agree with the theory that decline of oaks increases as trees age (Mueller-Dombois 1992).

Because the FIA damage data required considerable investigation and interpretation, we consider it of moderate value in measuring oak health. Starkey and others (1992) used some subjectivity in the identification of oak decline plots in the Southeast USA using FIA data. In our study the assignment of the various damage codes to "decline-associated" versus "all other" damage was somewhat arbitrary for several reasons, including the all-encompassing nature of oak decline and the interpretation required by the FIA field crew in assigning one damage code to a tree.

Based on LCR as a measure of oak health, more than 67 percent of the oak trees were in relatively good condition (LCR >30 percent) in both inventories. Furthermore, based on individual oaks measured in both inventories, 80 percent had either increases or no change in this attribute between inventories. This is the only variable for which change in condition of individual trees between cycles could be examined.

The most frequent LCR increases were in the smallest trees, which is expected because young trees divert proportionally more photosynthate to the crown than do older, larger ones (Kozlowski and others 1991). We consider LCR the variable least directly correlated to oak health. By itself, LCR is a very coarse measure of tree crown condition because it is based on the percentage of the total tree height that supports live green foliage and thus can vary in a forest stand because of tree density, adjacency to openings, and other factors unrelated to tree health. Compared to mortality, LCR may be more susceptible to observer variation (Ghosh and others 1995), although the three LCR classes we used in this study were quite general.

In summary, this preliminary study for Minnesota offers an approach to using FIA data to assess oak health. Mortality and damage data were found to be most useful measures of three variables considered and will be used to complete our regional historical assessment of oak health for the remaining six states in the Upper Midwest. The results of such analyses would complement information found in FIA forest inventory statistics publications for the states considered.

ACKNOWLEDGMENTS

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CLIMATE-TREE GROWTH MODELS IN RELATION TO LONG-TERM GROWTH TRENDS OF WHITE OAK IN PENNSYLVANIA

D.D. Davis and R.P. Long[1]

ABSTRACT.—We examined long-term growth trends of white oak by comparing tree-ring chronologies developed from an old-growth stand, where the average tree age was 222 years, with a second-growth stand where average tree age was 78 years. Evaluation of basal area growth trends suggested that an anomalous decrease in basal area increment trend occurred in both stands during the 1950s. To determine whether climatic factors could explain this decrease we developed conventional and ridge regression models based on step-wise selection of mean monthly temperature and total monthly precipitation variables for the period 1895 to1940. The years from 1941 to 1950 were used for verification of developed models; only the ridge regression models produced verifiable models. All models failed to predict actual growth in the period from 1951 to 1980 for both stands. The inability to predict growth response in the more recent period may relate to a non-constant climatic response of white oak or to non-climatic factors such as air pollution or altered atmospheric CO_2 concentrations that have affected tree growth-climate relationships.

The oak-hickory forests of Pennsylvania encompass 3 million ha of commercial forest land (Powell and Considine 1982). These forests are the dominant vegetation of the ridge and valley section of central Pennsylvania where species such as northern red oak (*Quercus rubra* L.) and white oak (*Q. alba* L.) are two of the most commercially valuable components (Powell and Considine 1982).

Various studies have documented that air pollution may reduce growth rates of trees growing near point sources of pollution (Long and Davis 1999, McClenahen and Dochinger 1985), even over large scales (Nojd and Reams 1996). Speculation that atmospheric pollutants or acidic deposition may adversely affect the tree growth without the appearance of external symptoms has been proposed in relation to some studies (Woodman and Cowling 1987, Phipps and Whiton 1988, McClenahen 1995). McLaughlin and others (1983) suggested that northern red oak growth decreased in some areas of northeastern U.S.A. due to increased air pollution. They based this conclusion on comparison of mean radial increment in the 1931 to 1955 period with growth in the 1956 to 1980 period. However, their study did not assess whether the growth decline was related to other factors such as tree age or climate.

Other studies have implicated atmospheric pollutants as factors that may alter or decouple tree growth-climate relationships (McClenahen and others 1999), resulting in conditions more limiting to tree growth (Puckett 1982, McClenahen and Dochinger 1985). In addition, the response of old-growth forests compared with second-growth forests to concurrent pollutants and climate may be quite different. This study was initiated to determine if white oak growth response to climate has changed since the 1950s, and to compare the growth response of old- vs. second-growth white oaks during the same time period. This study is not intended to be a comprehensive assessment of tree growth-climate processes, but is instead intended as a case study to develop approaches and analytical methods that facilitate analysis of these processes.

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INCREMENT CORE COLLECTION

An old-growth white oak stand and a second-growth white oak stand, growing approximately 17 km apart, were located in central Pennsylvania on and near Leading Ridge, Huntingdon County (table 1). Two, and sometimes three, 5 mm diameter increment cores were taken at breast height (BH, 1.4 m) from dominant and codominant trees in each stand (old growth n = 33; second-growth n = 30). Cores were air-dried, glued to grooved wooden holders, sanded to enhance the appearance of ring boundaries, and visually cross-dated (Stokes and Smiley 1968).

To assure accuracy of assigned dates, crossdating was verified using the computer program COFECHA (Holmes 1983, 1985). Ring-widths were measured to the nearest 0.01 mm on a Bannister Incremental Measuring Machine (Fred C. Henson Co., Mission Viejo, CA) interfaced with the microcomputer program TRIMS (Tree Ring Incremental Measuring System) developed by Madera Software (Tucson, AZ). Crossdating and/or measurement errors were identified and corrected. If the observed errors could not be corrected, the core was omitted from the chronology.

TREE-RING ANALYSES

Several methods were used to examine the relationship between growth trends and possible declines using the raw ring-width time series. To examine the non-climatic growth trend of individual trees, the basal area increment (BAINC) was calculated for each tree (Phipps 1983, Phipps and Field 1988). A cubic spline function with a 50 percent frequency response equal to 100 years was fit to the BAINC values of each core (Cook and Peters 1981). The resulting trend values of this curve were assumed to represent the growth trend of the tree without the influence of high frequency year-to-year climatic variation (Phipps 1983). The BAINC trend values were averaged for individual trees in each stand, and for the stand as a whole, and plotted to determine any changes in growth trends. BAINC approximates a straight line for most trees in the forest canopy, and a change in the slope of the BAINC may be evidence of endogenous or exogenous disturbance (Phipps 1983).

Table 1.—*Site characteristics and chronology statistics of white oak tree-ring chronologies developed from a second-growth and an old-growth stand in central Pennsylvania*

	Old-growth	Second-growth
Site characteristics		
Latitude (N)	40°40′	40°41′
Longitude (W)	77°45′	77°57′
Elevation (m)	363	442
% slope	10	5
Chronology statistics		
Chronology (yrs)	1562-1983	1876-1984
Number of trees	33	30
Number of radii	70	63
Mean sensitivity	0.175	0.170
Serial correlation	0.381	0.221
Standard deviation	0.221	0.175
Average age (yr)	222	78

Alternatively, to examine potential climatic factors affecting tree growth, non-climatic variance due to age trend and differential growth rates was removed by standardization. Raw ring-width values were divided by the value on a fitted line or curve (similar to the trend line that was used above) to provide a dimensionless tree-ring index (Fritts 1976). Ring-width data from the old-growth stand were double detrended with a negative exponential curve or linear regression line followed by a stiff cubic spline fit successively to the ring-width data (Cook and Holmes 1985). To limit the removal of climatic information from second-growth trees, ring-width data were detrended using a stiff cubic spline with a 50 percent frequency response of 100 years. Detrended, standardized ring-width indices were then averaged together by year into a robust mean value standard chronology for each stand (Cook and Kairiukstis 1989).

The standard chronologies from both stands were further characterized by calculating the mean sensitivity, first-order autocorrelation (or serial correlation), and standard deviation (Fritts 1976). Mean sensitivity, a measure of the ring-width index variation from one year to the next over the length of the chronology, was estimated by calculating the average of twice the absolute difference between adjacent indices divided by the mean of the two indices. First-order autocorrelation, an estimate of non-randomness in the ring-width series, was calculated as the correlation between year t and t-1 for all years in a chronology (Fritts 1976).

GROWTH-CLIMATE MODELS

The South Central Mountains Pennsylvania Climatic Division data, consisting of mean monthly temperature and total monthly precipitation averaged from all available single station records in the division for the period 1894 to

1980 was assembled and used to predict ring index growth (U.S. Environmental Data Service 1894-1984). Divisional data were used because only limited single station data were available from stations closest to the sampled areas. Because of the lag response of tree growth to climate (Fritts 1976), the climatic predictor variables were arranged in a 17-month dendroclimatic year beginning in May of the previous year and ending in September of the current year. This resulted in a pool of 34 candidate predictor variables for subsequent regression analyses.

Preliminary analyses indicated that the old-growth and second-growth chronologies were highly correlated ($P < 0.01$) with each other and with climate variables. Because of this correlation and to increase climate signal, the chronology ring-width indices from the old-growth and the second-growth stands were averaged to obtain a master chronology. This chronology was used for development of climatic models using stepwise regression for the calibration period from 1894 to 1940. The probability level for a climatic variable to enter and remain in the model was 0.15. The climatic data from 1941 to 1980 were used for model verification, and to determine whether growth response to climate was altered during this period.

Autocorrelation in tree-ring chronologies may obscure identification of climatic signal (Meko 1981). To limit the influence of the autocorrelation structure on derived climatic models, the master chronology was prewhitened with an order-2 autoregressive (AR) model for the period from 1730 to 1950. Prewhitening produces a white noise residual series by removing autocorrelation. The coefficients of the model were then used to prewhiten the data from 1951 to 1980. This procedure prevents the removal of any anomalous decline signal in the 1951 to 1980 time period, and assumes that the AR model describing the persistence structure from 1730 to 1950 is a time-stable representation of the autocorrelation structure of the chronology (Cook 1985). The AR coefficients were 1 = 0.246 and 2 = 0.197 with the model selected on the basis of the minimum AIC selection criterion (Akaike 1974).

For the climatic models developed in this study, both chronologies, one with the existent autocorrelation (unwhitened) and one produced by prewhitening (prewhitened) were used for all analyses. We did this because Monserud (1986) reported that hypothesis tests associated with unwhitened chronologies (those with existent autocorrelation left in the chronology) might be biased.

Multicollinearity, or linear dependencies among climatic variables, may also prevent identification of a reliable model, or lead to estimated regression coefficients which are too large in absolute value due to large variances and covariances of the least squares estimates of regression coefficients (Cropper 1985). To remedy this, ridge regression was used to produce biased coefficient values with smaller, though biased, standard errors. Ridge regression produces coefficient estimates after adding a small biasing parameter (k) to the diagonal elements of the $X'X$ matrix. This produces an inverse matrix ($X'X^{-1}$) in which the diagonal elements are less inflated (Cropper 1985).

Empirical evaluation of simulated climatic and tree-ring data suggested that a bias value of $k = 0.4$ would provide verifiable climatic response models (Cropper 1985). The ridge-biasing parameter for our study was selected after examination of the ridge trace plots of all 34 potential predictor variables (Montgomery and Peck 1982). Successive iterations of regression coefficients were calculated with the biasing factor beginning at 0 and continuing in 0.05 intervals to 1.0, in order to determine the minimum bias that resulted in reduced variance inflation factors (VIFs) (Montgomery and Peck 1982).

VIFs in excess of 5 to 10 indicate that associated regression coefficients are poorly estimated (Montgomery and Peck 1982). Because the VIFs in the data for the present study were slightly higher than those in the simulation studies, the ridge trace of all variables was examined to determine the bias where regression coefficients stabilized (Cropper 1985). The bias selected for our study was $k = 0.4$, based on both the ridge trace plots and the associated decrease in the VIFs. Stepwise regression of the biased climatic variables was performed as described previously with the unwhitened and prewhitened chronologies.

To determine whether tree growth response to climate has been altered since 1950, four climatic models (1 = unwhitened ordinary least squares (OLS) regression, 2 = unwhitened ridge regression, 3 = prewhitened OLS regression, 4 = prewhitened ridge regression) were first tested using successive 10-year verification periods from 1941 to 1980. The verification period predates the period of the suspected intervention of other growth-limiting factors so that model

stability can be evaluated prior to the intervention (Cook 1985). Pearson product-moment correlations between predicted indices and actual indices were calculated for each 10-year period, along with the associated probability level of the correlation.

A one-sided t-test was used to test whether the mean difference between actual and predicted indices (x) was significantly different from zero for each verification period. The reduction of error statistic (RE) was also evaluated (Lorenz 1956). This statistic has no associated probability level but is a rigorous verification statistic strongly affected by few deviations from observed values (Fritts and others 1979). RE values greater than zero that are accompanied by statistically significant product-moment correlations indicate that there is some predictive skill with the associated model (Cook 1985).

RESULTS

Plots of the stand-averaged BAINC trend with year-to-year climate fluctuations removed (fig. 1) indicate a slight decrease in the positive slope of BAINC within both the old-growth and second-growth stands. A linear regression line was calculated for the period 1870 to 1950 for the old-growth stand and used to estimate the predicted straight-line BAINC growth to 1980. The predicted straight-line growth was similarly estimated for the second-growth stand based on a linear regression for the 1920 to 1950 period.

For both stands, there is an apparent decrease in the BAINC trend starting in the 1950s. Because of the limited sample size in this study and the increase in standard deviation of the observed BAINC values, this predicted straight-line growth is within the 95 percent confidence interval of the actual BAINC curves. However, the consistent and simultaneous response of both the old growth (mean age = 222 years) and the second-growth (mean age = 78 years) stands suggests some non-climatic or low frequency climatic factor may be influencing tree growth.

Growth-Climate Models

Climatic models were developed to explore whether this decrease in growth was related to climatic variation, and to determine if growth response to climatic factors was altered during the 1950 to 1980 period. Because the chronologies were highly correlated, a master chronology was created by using the weighted average of the chronologies from the old- and second-growth stands (table 1). The master chronology mean sensitivity was 0.177, first-order autocorrelation was 0.417, and standard deviation was 0.206 for the chronology period 1562 to 1984.

The unwhitened and prewhitened chronologies (fig. 2, see sppendix) showed no evidence of a sustained decrease in tree growth in the 1940 to 1980 period, though evidence of the severe mid-1960s drought was indicated by below-average growth during this time period.

The coefficients of the climatic variables selected by stepwise regression to predict the unwhitened chronology indices are presented in table 2. For this model, nine variables were selected. The large standardized regression coefficients for the previous year's August precipitation (0.468) and the current year's June precipitation indicate the influence of these variables on tree growth. The fractional variance accounted for by the regression model (R^2) in the 1895 to 1940 calibration period was 0.714. The climatic model developed for the prewhitened chronology differed in variable selection from the model developed for the unwhitened chronology.

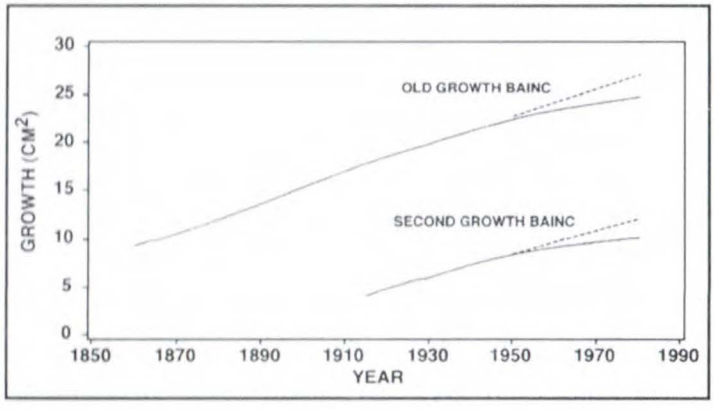

Figure 1.—*Actual (solid line) and predicted (dashed line) growth in terms of basal area increment (BAINC, cm²/yr) trends within old- and second-growth stands.*

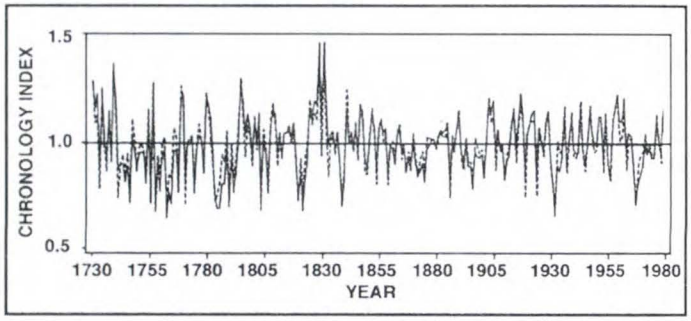

Figure 2.—*Unwhitened (solid line) and prewhitened (dashed line) standard master chronology derived by averaging the old- and the second-growth chronologies into a single master chronology.*

Table 2.—*Stepwise regression results for the unwhitened and pre-whitened chronologies. The standardized regression coefficients (b) and their standard errors (se) are given for the variables that entered the model. The probability level to enter and remain in the model was 0.15. The fractional variance explained by the model (R^2) and the R^2 adjusted for lost degrees of freedom (R^2 adjusted) were also calculated.*

	CHRONOLOGY TYPE			
	Unwhitened		Prewhitened	
Variable	b	se	b	se
Previous June temperature	-0.269	0.093	—	—
Previous June precipitation	—	—	0.196	0.101
Previous July precipitation	0.195	0.109	—	—
Previous Aug. precipitation	0.468	0.096	0.491	0.097
Previous Sept. precipitation	0.154	0.099	0.213	0.102
Previous Nov. temperature	0.341	0.101	0.418	0.099
Current Jan. temperature	—	—	0.208	0.102
Current Feb. temperature	0.172	0.100	0.179	0.105
Current April temperature	0.360	0.103	0.288	0.105
Current June precipitation	0.420	0.100	0.402	0.102
Current Sept. precipitation	-0.337	0.114	-0.342	0.104
R^2	0.714		0.704	
R^2 adjusted	0.643		0.630	

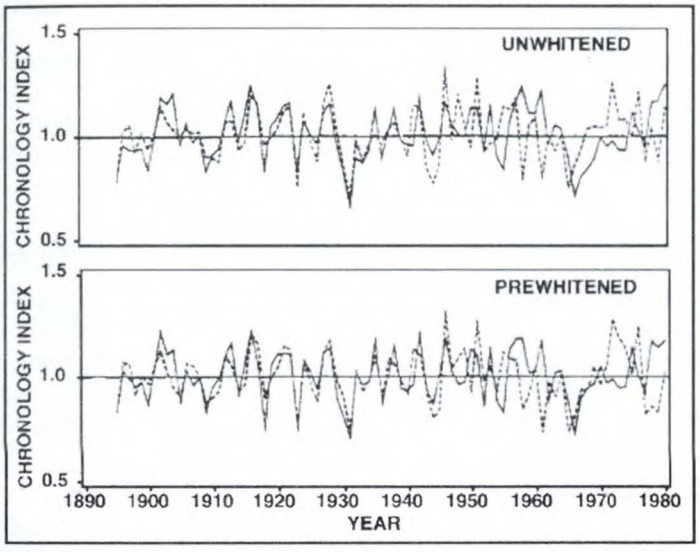

Figure 3.—*Actual (solid line) and predicted (dashed line) indices for the unwhitened and prewhitened chronologies calculated with ordinary least squares (OLS) regression models.*

The actual and predicted indices are plotted in figure 3. The previous year's June precipitation and current year's January temperature entered into the model, whereas the previous year's June temperature and previous year's July precipitation did not enter. The largest standardized regression coefficients were for the previous year's August precipitation (0.491) and for the previous year's November temperature. The R^2 for the model in the calibration period was 0.704, slightly lower than the R^2 for the model associated with the unwhitened chronology (table 2).

The verification statistics for these models are presented in table 3. For the model predicting the unwhitened chronology indices in the 1941 to 1950 period, the correlation (r) was 0.547, significant at P = 0.07. However, the reduction of error statistic (RE) was -1.335, indicating that the model did not have predictive capabilities. The mean difference between actual and predicted indices (x) was not significantly different from zero for this time period. The three subsequent verification periods did not indicate any positive correlation between predicted and actual unwhitened indices, or any predictive ability for this model as evidenced by the negative RE statistics. This model was only weakly verifiable in the 1941 to 1950 period, and had no ability to predict tree growth in the other three periods.

Verification of the prewhitened climatic model in the 1941 to 1950 period was even weaker, with the correlation (r) between actual and predicted indices (0.507) being significant at p = 0.133. The RE statistic was -0.511, indicating a lack of predictive ability for this model, and the mean difference between actual and predicted indices (x = -0.002) was not significantly different from zero. In the other three verification periods the correlations were either negative or close to zero, whereas the RE statistic remained negative. As with the model predicting the unwhitened indices, the verification is poor in the 1941 to 1950 period, while the model is unverifiable in all subsequent periods.

Ridge Regression

The diagonal elements of the X'X matrix, also known as the variance inflation factors (VIFs), were examined to determine if there was substantial multicollinearity or linear dependence among the candidate predictor variables. For the 34 climatic variables in the 1895 to 1940 calibration period, the VIFs varied from a minimum of 1.88 for the current year's April

Table 3.—*Pearson product-moment correlation (r), mean difference between actual and predicted indices (x), and the reduction of error (RE) statistic for verification of unwhitened and pre-whitened climatic models. Probability levels for r and the probability that x is significantly different from zero are indicated by asterisks (* < 0.15, ** < 0.10, *** < 0.05). No probability level is associated with RE; however values > 0 indicate substantial predictive capabilities for the model.*

Verification period	N	Model type: Unwhitened r	Unwhitened x	Unwhitened RE	Prewhitened r	Prewhitened x	Prewhitened RE
1941-1950	10	0.547**	0.011	-1.335	0.507*	-0.002	-0.511
1951-1960	10	-0.079	0.009	-1.336	0.064	0.002	-0.603
1961-1970	10	-0.111	-0.001	-0.706	0.038	0.048	-0.900
1971-1980	10	-0.116	-0.017	-1.424	-0.401	-0.004	-3.904

precipitation to a maximum of 8.91 for the current year's June precipitation. For the ridged climatic data, the VIFs varied from a minimum of 1.25 to a maximum of 1.87, indicating little or no multicollinearity among the candidate predictor variables.

The results of the stepwise selection of the ridge-biased climatic variables to predict the unwhitened chronology indices are presented in table 4. For the unwhitened chronology an 11-variable model was selected. Seven of these variables were also selected using the ordinary least squares (unbiased) regression. All estimated regression coefficients were lower than those estimated using ordinary least squares regression, a result of the shrinkage created by adding the bias. The largest coefficients were the previous year's August precipitation (0.240) and the current year's September temperature (-0.211). The fractional variance explained by the model was 0.544.

The stepwise model selected using ridged climatic variables to predict the prewhitened chronology indices was very similar to the model for predicting the unwhitened indices (table 4). The actual and predicted indices are plotted in figure 4. Eleven variables entered the model and those with the largest absolute coefficients were the previous year's August precipitation (0.263) and the previous year's November precipitation (0.213). As with the ordinary least squares regression model, the prewhitened prediction model included the previous year's June precipitation and the current year's January temperature variables. This model omitted the previous year's June temperature and previous year's July precipitation variables, which were included in the unwhitened prediction model. The fractional variance explained by this model was 0.529.

Table 4.—*Stepwise ridge regression results using climatic variables with a 0.4 bias to predict the growth of the unwhitened and prewhitened chronologies. The standardized regression coefficients (b) and their standard errors (se) are given for the variables which entered the model, along with the fractional variance explained by the model (R^2) and the R^2 adjusted for lost degrees of freedom (R^2 adjusted).*

	CHRONOLOGY TYPE			
	Unwhitened		Prewhitened	
Variable	b	se[a]	b	se[a]
Previous June temperature	-0.017	0.0068	—	—
Previous June precipitation	—	—	0.109	0.0002
Previous July precipitation	0.129	0.0003	—	—
Previous August precipitation	0.240	0.0002	0.263	0.0002
Previous Sept. precipitation	0.149	0.0002	0.189	0.0002
Previous Oct. temperature	0.137	0.0058	0.135	0.0056
Previous Nov. precipitation	0.177	0.0003	0.213	0.0002
Current Jan. temperature	—	—	0.164	0.0032
Current April temperature	0.185	0.0059	0.162	0.0058
Current June temperature	-0.189	0.0068	-0.191	0.0067
Current June precipitation	0.168	0.0002	0.156	0.0002
Current July precipitation	0.113	0.0002	0.129	0.0003
Current Sept. temperature	-0.211	0.0060	-0.199	0.0060
R^2	0.544		0.529	
R^2 adjusted	0.472		0.411	

[a]The standard error estimates are biased and ordinary inference procedures are not applicable because the distributional properties are not known.

The verification statistics for the stepwise ridge regression model indicate that the predicted unwhitened chronology indices were positively correlated with the actual indices in the 1941 to 1950 verification period (r = 0.654, p = 0.04, table 5). Coupled with the positive RE statisitic (0.145), this indicates that the model has predictive capabilities. The mean difference between actual and predicted indices (x = 0.022) was not significantly different from zero.

The remaining three verification periods indicated that the model could not predict tree growth during these periods (table 5). For 1951 to 1960, and 1961 to 1970, r was negative or close to zero. In the 1971 to 1980 period, r increased to 0.184, but the RE statistic was -0.206. The mean difference between actual and predicted indices (x) was not different from zero for any of the verification periods.

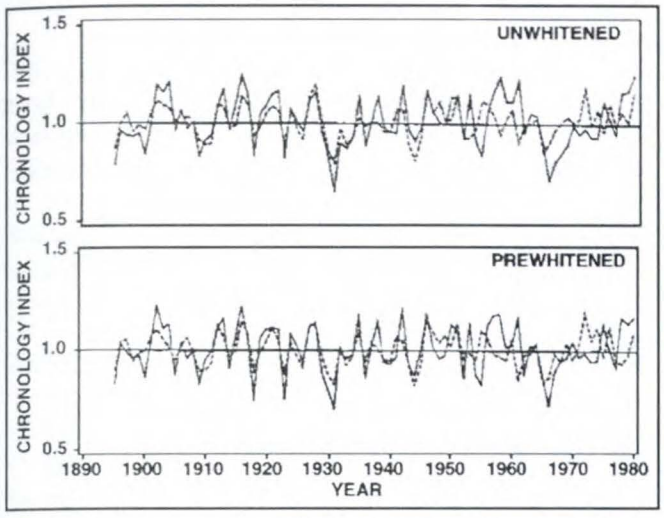

Figure 4.—*Actual (solid line) and predicted (dashed line) indices for the unwhitened and prewhitened chronologies, with predicted indices calculated with the ridge regression models.*

For the stepwise ridge model developed to predict the prewhitened chronology indices, the verification statistics were similar to those for the unwhitened chronology (table 5). The product-moment correlation (r) was 0.591 and the RE statistic was 0.254, indicating predictive skill for the model during the 1941 to 1950 verification period. The model was not verifiable in any of the subsequent three verification periods since correlations were not significant and RE statistics were negative. The mean difference between the actual and predicted indices (x) was not significantly different from zero during any period.

DISCUSSION

The apparent decrease in the non-climatic BAINC growth trend of both the old- and second-growth white oak stands suggests that an anomalous growth decline may have occurred. However, before the cause of this decline can be identified, other factors must be considered and a much larger sample size should be used. Some chestnut oak and white oak stands in northern Huntingdon County suffered high levels of mortality in the 1950s and 1960s from attacks of the pit-making scale, *Asterolecanium minus* (Ratz.), and some weakened trees were subsequently attacked by the two-lined chestnut borer (*Agrilus bilineatus* (Web.)) (Nichols 1968).

No specific stand history was available for the second-growth stand sampled in this study; however, it is possible that such insect attacks could account for the decrease in BAINC in this stand. Drought (McIntyre and Schnur 1936, Jenkins and Pallardy 1995), climate (Tainter and others 1990), diseases (Fergus and Ibberson 1956), and other factors (McClenahen and others 1997) also may cause oaks to decline and resultant growth symptoms to persist for many years (Staley 1965, Biocca and others 1993).

Table 5.—*Pearson product-moment correlation (r), mean difference (x) between actual and predicted indices, and the reduction of error (RE) statistic for verification of unwhitened and pre-whitened ridge regression climatic models. Probability levels for r and the probability that x is significantly different from zero are indicated by asterisks (* < 0.15, ** < 0.10, *** < 0.05). No probability level is associated with RE; however values > 0 indicate substantial predictive capabilities for the model.*

Verification period	N	MODEL TYPE: Unwhitened r	Unwhitened x	Unwhitened RE	Prewhitened r	Prewhitened x	Prewhitened RE
1941-1950	10	0.654***	0.022	0.145	0.591**	0.009	0.254
1951-1960	10	-0.073	0.036	-0.564	0.044	0.018	-0.247
1961-1970	10	0.048	-0.029	-0.209	0.144	0.008	-0.200
1971-1980	10	0.184	-0.013	-0.206	-0.305	-0.005	-1.256

Similarly, the advanced age of trees in the old-growth stand could account for the observed decrease in BAINC. However, examination of the individual tree BAINC curves from the old-growth stand did not reveal a consistent association of the older tress (tree > 230 years old) with decreasing BAINC. Other stand-wide disturbances including, but not limited to, such factors as insect defoliations or disease epidemics, must by considered before assessing the role of atmospheric pollutants.

Growth-Climate Models

The unwhitened and prewhitened standardized chronology indices (fig. 2) provide no evidence of any sustained decline in ring-width growth since the 1950s. Analysis of climatic models indicated that models developed using ordinary least squares (OLS) regression were weakly verifiable in the 1941 to 1950 period, and unverifiable in any subsequent periods.

Several factors either alone or in combination may prohibit identification of a verifiable climatic model (Cook 1985). First, the 34 candidate predictor variables may not contain the variables necessary to accurately describe and predict tree growth. Secondly, the screening of the predictor variables may have been confounded by multicollinearity, or the selected model(s) may have been spurious since the number of candidate predictor variables was a large fraction of the number of observations. Thirdly, the tree-ring time series may not contain a stationary, time-stable climatic signal. Also, the intervention period when growth was altered may actually precede the hypothesized 1950 to 1980 period. Finally, a stand-wide disturbance may have affected tree growth during the verification period. To prevent this latter problem the two individual chronologies were averaged into a master chronology, thus limiting the influence of stand disturbance(s).

Because of the large number (34) of candidate predictor variables, multicollinearity may have affected variable selection and prevented verification of the OLS stepwise regression models. Stepwise ridge regression was used to ameliorate this multicollinearity problem. Using simulation studies with known inputs and outputs, Cropper (1985) observed that ridge regression more frequently and accurately estimated known coefficients of climatic response function variables than either OLS multiple linear regression or principal components regression. While Cropper's analyses used simulated data, the results indicated that ridge regression could adequately account for multicollinearity and limit its influence on derived climatic models.

In the present study both ridge regression models (using unwhitened and prewhitened dependent variables) were verifiable in the 1941 to 1950 period. Both models predicted tree growth that was significantly correlated with actual growth, and both models had positive RE statistics, indicating that they had predictive skill during this verification period. This suggests that the ridge regression models derived from the 1895 to 1940 calibration period were a time-stable representation of growth response to climate. The failure of both ridge regression models to predict tree growth in the 1950s indicates a possible change in tree growth response to climate.

Various hypotheses have been suggested to explain such a change (Puckett 1982, Cook 1985, McClenahen and Dochinger 1985). If the tree-ring time series used for climatic modeling in this study did not contain a stationary climatic signal, then no model would be successful in describing tree growth. The verification of the ridge regression models in the 1941 to 1950 period refutes this explanation; however, climatic change, or threshold climatic response could account for such a change in tree growth response to climate. A climatic variable such as drought may not significantly affect tree growth until it exceeds a critical threshold of severity (Cook 1985), and this threshold may differ depending on soil conditions at the microsite or individual tree level.

Anomalous climatic patterns have been documented throughout the northern hemisphere during the 1960s and 1970s. A series of extreme winter temperatures in the United States since 1975 has been shown to have a return time of 1,000 years (Karl and others 1984). Cook (1985) suggested that such a phenomenon might have caused increased susceptibility of red spruce in the northern Appalachians to winter freeze injury. The higher frequency of winter freeze damage since 1967 may have led to increased tree injury, decreased growth, and altered growth response to climatic variables (Cook 1985). In addition, several anomalous climatic events occurred in Pennsylvania during the 1950s, at which time an unexplained dieback and decline of oaks occurred in various areas within the northeastern United States.

Staley (1965) suggested that the decline could have been related to growing season moisture deficits in some years, but that drought was not the primary cause of the decline. Moisture deficits averaging 10 cm per year were recorded in Pennsylvania for the period from 1952 to 1955, and total annual precipitation was substantially below normal in 1957 (Nichols 1968). During the 1960s a prolonged drought affected the northeastern United States. The effects of this drought are notable in the chronology plots (fig. 2), with the most severe growth reduction occurring in 1965 or 1966.

Three successive severe winters occurred in Pennsylvania in 1976-77, 1977-78, and 1978-79, with the winter of 1976-77 the coldest ever recorded in Pennsylvania (Yarnal 1987). These climatic events may have exceeded a threshold so as to alter tree growth response to climate, and may be inadequately described by climatic models. Persistent climatic anomalies may limit tree growth and possibly trigger physiogenic forest declines (Hepting 1963).

The ridge regression climatic models indicate some effect of changed environmental conditions affecting tree growth during 1950 to 1980. McClenahen and Dochinger (1985) observed a similar change in tree growth response to climate near an urban-industrial complex in Ohio after the 1930s. Similarly, Puckett (1982) speculated that atmospheric pollutants might act alone or in synergy with other growth-limiting factors to alter the relationship of tree growth to climate. Such a phenomenon may have altered growth response of pitch pine, eastern white pine (*Pinus strobus* L.), and chestnut oak (*Q. prinus* L.) to climate in southeastern New York (Puckett 1982).

Trees sampled in the present study were remote from any significant point sources of air pollution and from large urban-industrial complexes. However, other air pollutants such as ozone may be transported long distances and may adversely affect remotely located forest trees (Hayes and Skelly 1977). Similarly, the trees sampled for the present study receive precipitation with an average pH of approximately 4.0 along with substantial inputs of sulfates and nitrates (Lynch and others 1985). It is unknown whether these factors could alter growth response to climate. The absence of long-term atmospheric monitoring data at remote locations limits the ablility to draw conclusions about changes in these parameters.

Based on the ridge regression climate models developed for this study, a sustained decline in white oak growth is inconsistent both with model predictions and with actual growth during the 1950 to 1980 period. However, substantially altered growth response to climatic variables is indicated by the absence of model verification in this period. Other growth-limiting factors are apparently modifying tree growth response to climate. Anthropogenic pollutants, climate change, or other biotic and abiotic factors mentioned above may play a role in altering growth response to climate. More spatially extensive sampling will be necessary to rigorously test these relationships before causal factors can be identified.

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APPENDIX

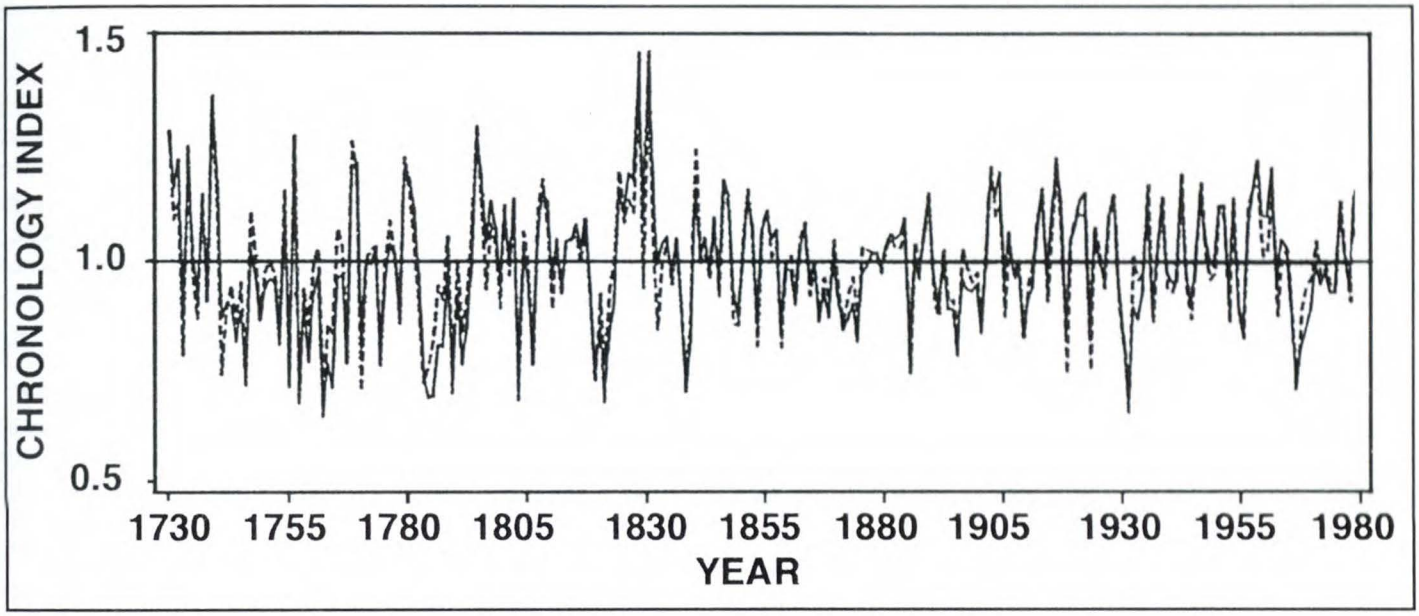

Figure 2.—*Unwhitened (solid line) and prewhitened (dashed line) standard master chronology derived by averaging the old- and the second-growth chronologies into a single master chronology.*

GROWTH AND MORTALITY OF BIGTOOTH ASPEN TREES STRESSED BY DEFOLIATION

Donald D. Davis and Timothy M. Frontz[1]

ABSTRACT.—A survey conducted by the authors in 1993 in six stands within an area of declining aspen revealed that aspen mortality ranged from 25 to 67 percent per stand. Tree-ring analyses revealed that trees dead at time of sampling in 1993 had been growing slower in four of the six stands during the previous decade than were the surviving trees. However growth declines had also occurred in trees that were alive in 1993. The greatest mortality of bigtooth aspen trees occurred in 1992, following 2 consecutive years of severe gypsy moth defoliations. Based on basal area decline and stand deterioration in response to defoliation, we recommend harvesting bigtooth aspen stands in this area well prior to 70 years of age.

Bigtooth aspen (*Populus grandidentata* Michx.) is an important tree species managed for wildlife within even-aged, mixed-hardwood forests of northern Pennsylvania. However, local forestry personnel reported that canopy aspen trees in this area had suddenly declined in the early 1990s, a condition interfering with wildlife management plans. Following several field trips to the area and conversations with local forestry personnel, we theorized that the current decline was related to past stresses. Although aspen stands generally begin to deteriorate at 40 to 45 years on poor sites, they can be maintained for 60 to 70 years on good sites before deterioration and root rot becomes a serious problem (Laidly 1990).

The onset of mortality in the stands in question, which were of unknown age, was quite sudden. The temporal onset of such stresses may be inferred retrospectively by using tree-ring analyses (Fritts 1976, Cook and Kairiukstis 1989, McClenahen 1995, LeBlanc 1996) including the use of cross-dating (Stokes and Smiley 1968, Phipps 1985, Hart 1989), a procedure used to assign an exact year to a specified ring. Comparison of growth trends of healthy vs. declining populations of trees may provide insight as to when a current decline was initiated and possible causes (Tainter and others 1990). The objective of this investigation was to compare growth patterns of aspen that were live vs. dead at time of sampling.

METHODS

This study was conducted within even-aged, mixed-hardwood stands in the Tioga State Forest of northern Pennsylvania. Six stands, each of which contained at least 40 percent basal area of bigtooth aspen, previously had been selected to study the relationship between *Armillaria* (*sensu lato*) and dead and dying aspen (Frontz 1997, Frontz and others 1998). A 100 percent survey of bigtooth aspen trees within each of the six stands was conducted and the percentage of aspen mortality determined. Fifteen live and 15 dead bigtooth aspen trees within each of the six stands (n = 180) were randomly selected and felled in early May 1993 for tree-ring analysis. All sample trees were in the dominant or codominant crown class.

Stem sections, 20 cm in length and centered at 1.4 m above ground (breast height – BH), were removed from each sample tree. To minimize cracking, sections were air-dried for 5 weeks and then kiln-dried at progressively higher temperatures from 32 to 49 C for 7 weeks, at which time the stem sections had a moisture content of 8 percent. A 5-cm thick disk was cut from the center of each dry stem section, sanded with progressively finer grit sandpaper, and polished with lamb's wool to enhance ring boundaries.

Four radii, spaced approximately 90° apart, were penciled from the pith to the bark on each

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cross-sectional disk. Each disk was inspected along the marked surface for ring clarity, decay, cracks, discontinuous rings, and other obvious problem areas. The two radii judged to be least problematic were selected for ring-width measurements from the pith to the outside of the most recent growth ring. All individual tree-ring series were visually cross-dated (Stokes and Smiley 1968, Holmes 1983, Phipps 1985) and the entire circumference of every tenth ring was circumscribed on each disk to minimize measurement errors associated with discontinuous rings. Ring-widths were measured to the nearest 0.01 mm using a Bannister Incremental Measuring Machine (Fred C. Henson Co., Mission Viejo, CA) interfaced with the microcomputer program TRIMS (Tree Ring Incremental Measuring System) developed by Madera Software (Tucson, AZ).

Data were examined with the quality-control program COFECHA (Holmes 1985) to aid matching each ring to the correct year. Any chronologies that did not cross-date were eliminated from subsequent data analyses. Ten percent of the acceptable series were selected randomly for remeasurement. Any error between the original and remeasured series was compared, and if the error was greater than acceptable limit (Phipps 1985), the source of the error was located and corrected by re-measuring. This process was continued until measurements were within prescribed acceptable limits. Mean age was determined for each stand using only cross-dated series.

Linear ring-width and radius measurements for each tree were converted to basal area increment (BAI), the cross-sectional area (sq cm) of annual xylem produced per tree (Phipps and Field 1988). Annual values for individual trees in each stand were averaged to create a stand-average BAI, which was plotted and examined visually for relationship to known stress events. The relation of the growth series among the six stands was calculated using average series intercorrelation (Cook and Kairiukstis 1989).

Initial results from the tree-ring analysis indicated that radial growth in some stands had started to decline during the late 1960s. Therefore, the mean total BAI from 1968 to 1989 was calculated per stand and compared to determine if there were significant differences in total growth among stands during this period. Data were examined using analysis of variance and mean separation tests based on Tukey's pairwise comparisons ($P < 0.01$) (Phipps and Field 1988).

To aid in management decisions, the percentage of total BAI accumulated at various ages was compared among the six sites to help determine optimal harvest age. Also, the year in which each tree died was determined by noting the last year of growth on trees that were dead at time of sampling; year-of-death values were expressed as relative frequencies (Frontz 1997). In order to compare the growth of live vs. dead trees (at time of sampling), ratios of growth data were constructed and plotted for each year. The horizontal line indicates years in which both sets of trees were growing at the same rate. Values above the horizontal line (greater than "1") indicate that the trees that ultimately died were actually growing greater than the trees that were alive at time of felling.

RESULTS

Based on visual examination and COFECHA analysis, 84 percent of the tree-ring series could be cross-dated and therefore were used in further analyses. Growth patterns of both live ($n = 81$) and dead at time of sampling ($n = 71$) aspen trees generally exhibited synchronous variation. This finding indicated that events that resulted in wide or narrow growth rings were generally reflected in the growth patterns of both live and dead (at time of sampling) aspen trees in all six stands (fig. 1). Average series intercorrelation among the six sites was 0.59, being greatest at the Cushman site (0.69) and least at the Arnot 6 site (0.55).

Bigtooth aspen trees at the Deer Fence site were the youngest, averaging 69 years at BH. Trees in the Arnot 5, Arnot 6, Butler Hollow, and Cushman sites averaged 77 to 80 years. Aspen trees at the Seaman's Trail site were the oldest, having a mean age of 86 years. The incidence of tree mortality was greatest at the Arnot 5 site (67 percent) and least at the Seamans Trail site (25 percent). Approximately 90 percent of the dead sample trees had died between 1990 and 1992, and 50 percent of the dead trees had died during a single year, 1992.

Aspen trees in all six stands exhibited declining growth patterns in BAI during their last 25 to 40 years of growth, regardless of final state of health (fig. 2). In general, most trees in the Arnot 6 and Deer Fence stands started to decline in the mid-1970s, whereas trees in the other stands started a general decline in the early 1950s. Aspen trees that were dead at time of sampling in four stands (Arnot 6, Cushman, Deer Fence, and Seamans Trail) generally grew slower during the last *ca* 20 years of their lives

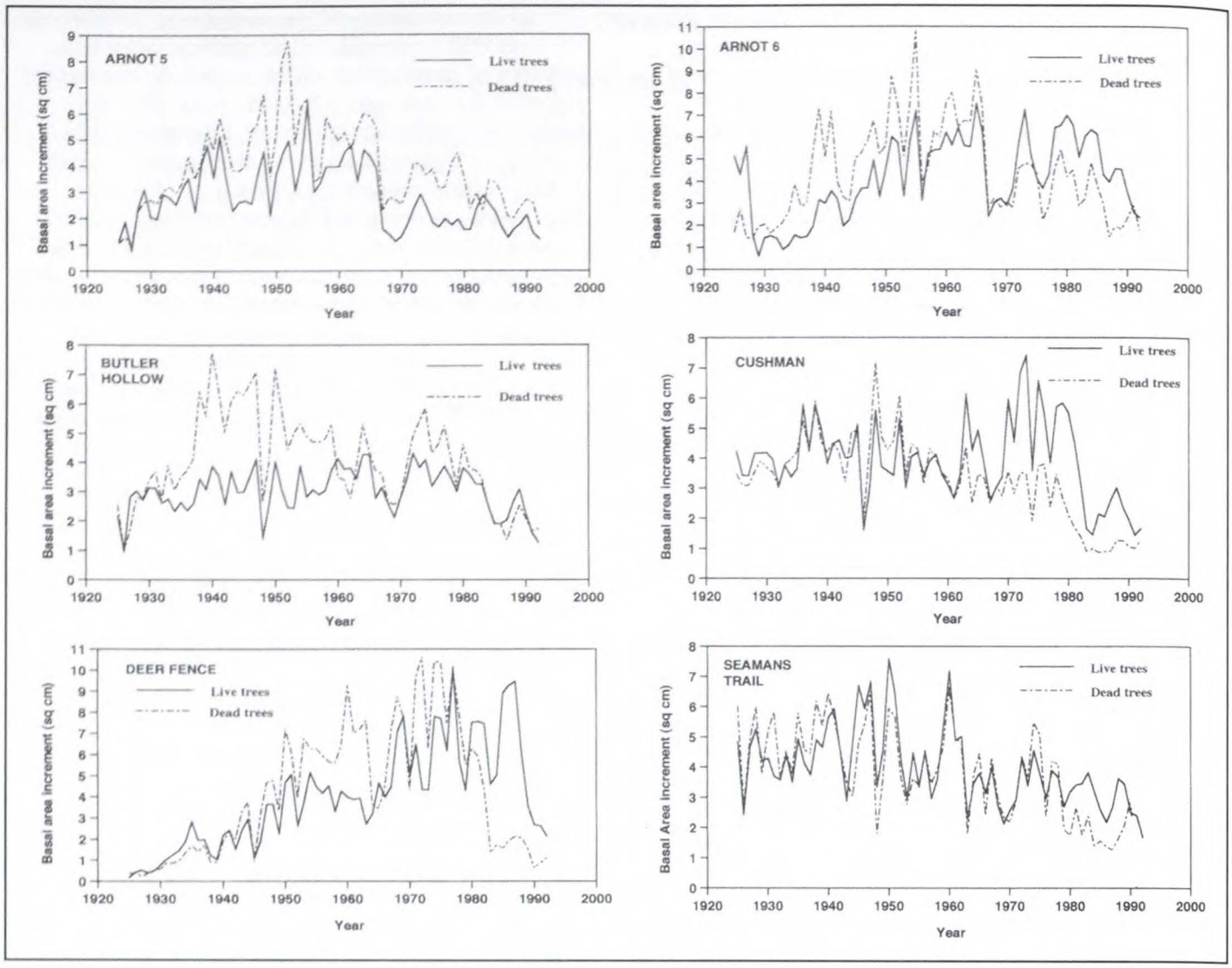

Figure 1.—*Solid lines indicate growth patterns of trees that were alive at time of felling and sampling in 1993. Dashed lines represent growth patterns of trees that were dead at time of felling in six stands in northern Pennsylvania.*

as compared to those trees that were alive at time of sampling. The growth rate displayed by aspen trees in Arnot 5 and Butler Hollow prior to dying often exceeded that of the trees that were alive at time of sampling. Trees that ultimately died in Arnot 5, Arnot 6, Butler Hollow, and Deer Fence stands often had greater growth rates in their early years than did those trees that were alive when sampled.

Trees that died in the Arnot 6 and Cushman stands exhibited declining growth beginning *ca* 1970 and generally continued to decline in growth until death. Trees that died in the Deer Fence and Semans Trail stands had a greater growth rate than the stand average until the mid-1970s and then declined until death.

Mean total BAI of sampled trees from 1968 to 1989 at the Deer Fence site (141.4 sq cm) and the Arnot 6 site (117.5 sq cm) were not statistically different, and were significantly greater than the mean total BAI of trees from the other four sites ($P < 0.01$). The mean total BAI of trees from the Arnot 5 (78.7 sq cm), Butler Hollow (80.2 sq cm), Cushman (78.7 sq cm), and Seamans Trail (80.6 sq cm) sites were not significantly different from each other ($P < 0.01$). Approximately 60, 75, and 92 percent of the total BAI from all six sites was produced in trees 50, 60, and 70 years of age, respectively.

DISCUSSION AND CONCLUSIONS

Tree-ring analyses were utilized in this study to investigate tree decline (McClenahen 1995, LeBlanc 1996). However, it has been our experience that bigtooth aspen often contains discontinuous rings, making it more useful to fell trees and take disks, as opposed to using increment borers (Trujillo 1975). Trees that were dead at

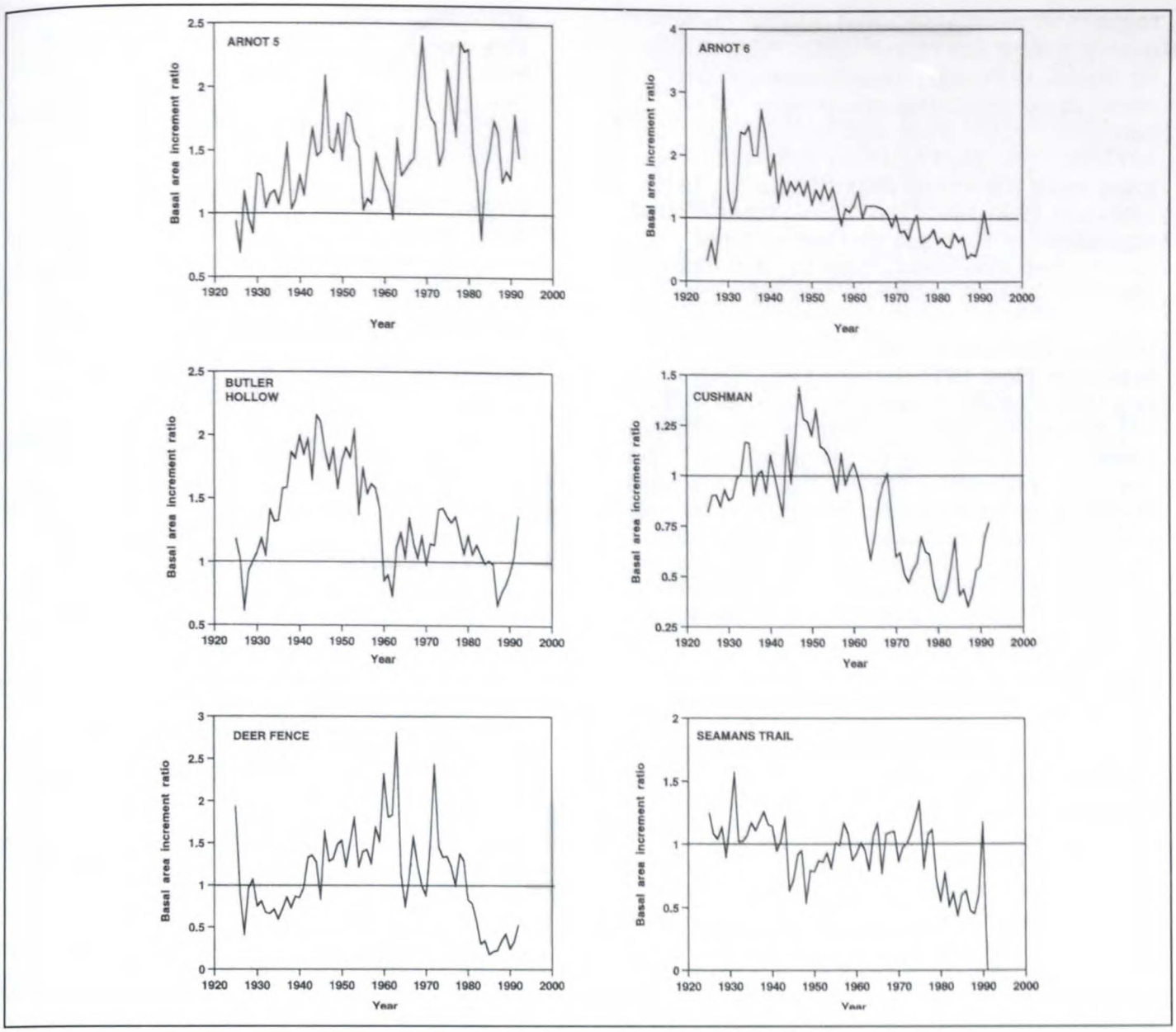

Figure 2.—*Ratio of Basal Area Increment (BAI) growth (sq cm) of bigtooth aspen trees in six stands in north-central Pennsylvania: The horizontal line at "1" indicates the value derived if the annual growth of the trees that were dead at time of felling equaled the annual growth of live trees (growth of dead trees/growth of live trees = 1). On the curve, values less than 1 indicate that the trees that were dead at time of harvest had been growing less than those trees that were alive at felling.*

time of felling in 1993 showed persistent patterns of growth decline in the decade prior to death in four of the six stands (fig. 1). Root excavations revealed *Armillaria* species associated with roots of dead and dying aspens (Frontz 1997, Frontz and others 1998).

In four of the six stands, trees that ultimately died were actually growing better in their early years than were trees that ultimately survived. This latter finding is similar to that reported by Jenkins and Pallardy (1995) for oak trees on relatively good sites, suggesting that rapid early growth rates may predispose trees to early death under certain conditions. Other studies have also reported that declines may follow periods of unusually high growth (Tainter and others 1984, VanDeusen and Snow 1991, Biocca and others 1993). Although faster growing trees have larger stem diameters, they may allocate less carbon to their roots, making them more susceptible to root diseases (Waring 1987, Jenkins and Pallardy 1995) such as Armillaria root rot (Wargo and Harrington 1991), or they may allocate less carbon to chemical defenses against insect attack (Waring 1987).

Bigtooth aspen is a preferred host for gypsy moth (Chilcote and others 1992). Unpublished PA Bureau of Forestry records indicate that severe insect defoliation events occurred on hardwoods in the study area in 1966-1967 and 1971-1972, followed by severe defoliation from gypsy moth (*Lymantria dispar* L.) during 1982-1983 and 1990-1991. Hart (1989) reported that aspen trees in Michigan that had suffered severe gypsy moth defoliations in 1986 died in 1987-1988 due to Armillaria root rot infections.

Different clones may respond differently to defoliation (Wall 1971, Barnes 1989, Chilcote and others 1992). Clonal differences, as well as site differences, may have influenced the differences in growth among the six stands. Both the live and dead (at time of sampling) trees in most stands exhibited reduced growth pulses, possibly related to insect defoliation (fig.1). In addition, the year-of-death peaked in 1992, immediately following severe defoliation by gypsy moth in 1990-1991. Again, the differential susceptibility of different aspen clones to stress may be responsible for the differences in mortality among the six stands.

Mean total BAI of sampled trees from 1968-1989 of trees within the Deer Fence and Arnot 6 stands were greater than that of the other stands. During related studies, Frontz (1997) noted that there were differences among stands in site factors such as soil stoniness and stand density. These were not quantified, but such differences, along with the clonal influences, may help explain differences in total growth among stands. Five of the six stands examined in this study were greater than 70 years old. Aspen trees at the Seamans Trail site were the oldest, averaging 86 years; however, stand mortality was the least (25 percent) at this site, indicating that mortality in these stands was not related to tree age. Nevertheless, 25 percent is still an unacceptable level of mortality.

In our study, 92 percent of the average total BAI for bigtooth aspen trees in the six stands had been achieved by age 70. This finding, along with the fact that aspen mortality in the six study stands ranged from 25-67 percent, allows us to recommend harvesting aspen stands in this area well prior to 70 years of age, perhaps even as early as 50 years.

ACKNOWLEDGMENT

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SURVIVAL OF OAK ROOT SYSTEMS FOLLOWING FRILL GIRDLE HERBICIDE TREATMENT FOR OAK WILT CONTROL

Johann N. Bruhn; James J. Wetteroff, Jr.; and Linda Haugen[1]

ABSTRACT.—Mechanical separation of root systems is widely used to prevent tree-to-tree vascular spread of oak wilt disease. A safe effective herbicide treatment would be valuable for this purpose in hilly, rocky, or urban settings. Three treatments were frill-girdle applied:

1) water,
2) undilutetd Garlon 3A (trichlopyr), or
3) half-strength aqueous Garlon 3A plus 24 ml per L Arsenal AC (imazapyr).

Autumn, winter, and late summer treatments were applied to white, northern red, and black oak. Root system mortality was evaluated with the vital stain 2,3,5-triphenyltetrazolium chloride. Root mortality occurred much more slowly than canopy mortality and too slowly for useful oak wilt control.

Disruption of inter-tree vascular connections via mechanical or chemical separation of grafted root systems is necessary to prevent tree-to-tree transmission of the oak wilt disease pathogen and halt expansion of oak wilt disease epicenters. Five foot deep vibratory plow line "barriers" to pathogen transmission are the currently favored tool for this purpose (for example, Blankenheim and others 2000). However, vibratory plows and 5 foot blades are not universally available and are difficult or unsafe to use in rocky soils, on steep terrain, or in urban settings. There is also concern regarding the durability of this kind of root graft barrier. For example, neither the likelihood nor the timing of root growth and re-grafting across plow lines are known.

Clearly, there is demand for an alternative tool for functional separation of grafted root systems. An herbicide treatment that safely and reliably kills oak root systems would be a particularly useful tool for halting epicenter expansion in locations inappropriate for the operation of a large tractor pulling a 5 foot blade. A variety of herbicides have been routinely used to kill unwanted trees. At least some of these herbicides can be translocated across root grafts to untreated trees, because herbicide damage to neighboring untreated trees has been observed.

The two herbicides which were most often recommended to the authors for their ability to kill red oak species were Arsenal AC (imazapyr) and Garlon 3A (trichlopyr). Because Arsenal AC is more soil-active than Garlon 3A, the possibility of non-target tree and turf damage is greater with Arsenal AC (Dr. James Miller and Max Williamson, personal communications). Aqueous mixtures of Garlon 3A and Arsenal AC have been used operationally by the USDA Forest Service, most prominently to kill unwanted hardwoods in the southern United States (Dr. James Miller and Max Williamson, personal communications). The seasons of application most often recommended were autumn (when the trees are translocating sugars downward for winter storage) and late winter (basis unknown). If herbicide treatment in the late summer (as soon as annual oak wilt symptom expression is complete) is efficaceous, epicenters could be treated when pathogen position is most accurately known.

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Our overall objective was to evaluate the effectiveness of two highly recommended herbicide treatments as potential management tools to block expansion of oak wilt epicenters where vibratory plow use is impractical. Our specific initial objectives were to:

1) identify set(s) of conditions (chemical and season of application) under which root systems of healthy susceptible trees might be quickly rendered unsuitable for service as root graft "bridges" for the expansion of oak wilt disease epicenters; and
2) determine the effectiveness of the selected treatments for killing tree root systems.

MATERIALS AND METHODS

Two hundred well-separated study trees were selected at each of three locations: Little Lost Creek Conservation Area (LLCCA) near Warrenton, MO (Warren Co.); Daniel Boone Conservation Area (DBCA) near Jonesburg, MO (Warren Co.); and Deer Ridge Conservation Area (DRCA) northwest of Hannibal, MO (Lewis Co.). The oak species under study were: *Quercus velutina* Lam. (black oak) and *Q. alba* L. (white oak) at LLCCA, *Q. velutina* and *Q. rubra* L. (northern red oak) at DBCA, and *Q. alba* and *Q. rubra* at DRCA. Each tree selected for study at each site was separated by at least 20 m from any other study tree. Larger trees were separated by greater distances from surrounding trees (see table 4 in Bruhn and others (1992) as a rough guide.

Treatments

One complete set of treatments was installed at each study site in Autumn (late-October/ early-November 1998), and another in late Winter (mid- to late-February 1999). A third complete set of treatments was installed at DBCA and LLCCA in late summer (mid-September 1999). Scheduling conflicts precluded establishment of late summer treatments at DRCA. Experimental factors were:

1) two oak species at each study site (see above);
2) two candidate herbicide treatments plus two relevant control treatments (see below); and
3) the two (DRCA) or three (LLCCA and DBCA) seasonal applications just indicated.

Ten trees per study species at each location were randomly assigned to each combination of experimental factors.

The two experimental herbicide materials used were:

1) Garlon 3A undiluted, and
2) half-strength aqueous Garlon 3A to which was added 24 ml Arsenal AC per liter of the diluted Garlon 3A.

This mixture was recommended by Max Williamson (personal communication). Both materials were prepared fresh prior to application and applied to frill girdles (FG), with cuts placed edge to edge, angled downward, and penetrating the xylem at least 1.25-cm. Trees were girdled approximately 1 m above ground level (Miller 1991). Herbicide was applied at the rate of 1 ml per 5 cm of stem circumference. The control treatments were:

1) FG, with tap water only; and
2) no FG and no herbicide.

At each study site, the same 10 trees per species served as "no frill girdle (FG) and no herbicide" reference controls in all three seasonal treatments.

It was initially intended to conclude each portion of the experiment 1 year after installation. However, observations of canopy condition and basal and stem sprouting indicated that the Autumn 1998 treatments had not yet caused adequate mortality (in light of our objective of complete tree mortality) by Autumn 1999 to warrant destructive sampling of lateral roots. Ultimately, the study was concluded in three stages: between 10-24 July 2000 at DBCA; between 11-14 June 2001 at LLCCA; and between 10-11 September 2001 at DRCA. In this manner, a single complete set of root vitality data was collected at a single point in time from each site. As a result, Summer treatments were allowed to develop for 7 and 10.5 months less, respectively, than Winter and Autumn treatments. Also, all treatments at DBCA were allowed to develop for 11 and 14.5 months less than the corresponding treatments at LLCCA and DRCA.

Tree Health Evaluation

Initial study tree condition (crown dieback, foliage transparency, and numbers of root crown and stem sprouts) was first evaluated for all study trees in October 1998 (Millers and others 1992). Study tree condition was similarly re-evaluated at study conclusion on each site; the condition of surrounding untreated trees was observed at the same time for indications of possible treatment influence. We define crown dieback as progressive branch mortality from the tip toward the bole, measured as the percentage of a tree canopy's total two-dimensional profile which is represented by branch dieback (see Millers and others 1992). We define and measure foliage transparency as the percentage of a tree's live canopy area through which sky is visible (see Millers and others 1992). We also report here the percentage of study trees in each treatment which supported live sprouts from the root crown during the final evaluation.

In addition, cambium vitality was evaluated on two lateral roots 5 to10 cm from their point of distinction from the root crown, either visually (based on obvious tissue discoloration or decay) or by use of the vital stain 2,3,5-triphenyltetrazolium chloride (TTC) (Parker 1953, Joslin and Henderson 1982). To evaluate a root, a block of root wood with one face representing approximately 0.5 cm by 4 cm of vascular cambium surface was immersed in a 1 percent aqueous solution of TTC in a glass screw-cap vial. The cambium surface of living samples turned red within an hour. Each evaluated root was considered to be a replicate in the sense of Glover (1991). One tree representing each species and treatment combination at DBCA was uprooted in late July 2000 to determine the health of lateral roots beyond the root crown using TTC.

Statistical Analysis

To identify the treatment factors which contributed to fastest root mortality, we used stepwise logistic regression analyses (SLR; see Sokal and Rohlf 1995), employing PROC LOGISTIC ($\alpha = 0.05$; SAS/STAT System Release 6.12, SAS Institue, Inc., Cary, NC). The response variable employed was the binary (classification) variable individual root mortality. The independent variables included were: the incubation period between treatment application and evaluation (a continuously distributed variable), and treatment season (Autumn, Winter, and Summer), FG treatments (none, water, Garlon 3A, and Garlon 3A + Arsenal AC), and tree species (*Q. alba*, *Q. rubra*, and *Q. velutina*).

RESULTS

All treatments at DBCA (*Q. rubra* and *Q. velutina*) were evaluated in July 2000, after 20.5 months, 17 months, and 10 months incubation, respectively, of the Autumn, Winter, and Summer applications (table 1). The canopies of all trees treated with Garlon (except one *Q. velutina* treated with Garlon in the summer) or the Garlon plus Arsenal mixture appeared dead. The canopies of trees which received the FG plus water treatment averaged at least 48 percent dieback and 64 percent canopy transparency, whereas untreated "control" *Q. rubra* and *Q. velutina* trees averaged, respectively, no more than 17 and 22 percent dieback and 12 and 18 percent canopy transparency. Of the trees receiving FG plus water, Garlon, and Garlon plus Arsenal, respectively, 68 percent, 25 percent, and 30 percent supported live sprouts at the root crown in July 2000.

All treatments at LLCCA (*Q. alba* and *Q. velutina*) were evaluated in June 2001, after 31.5 months, 28 months, and 21 months development, respectively, of the Autumn, Winter, and Summer applications (table 2). The canopies of all trees treated with Garlon or the Garlon plus Arsenal mixture appeared dead. The canopies of trees which received the FG plus water treatment averaged at least 88 percent dieback and 83 percent canopy transparency, whereas untreated "control" *Q. alba* and *Q. velutina* trees averaged, respectively, no more than 18 and 28 percent dieback and 17 and 26 percent transparency. Of the trees receiving FG plus water, Garlon, and Garlon plus Arsenal, respectively, 47 percent, 18 percent, and 23 percent supported live sprouts at the root crown in June 2001.

All treatments at DRCA (*Q. alba* and *Q. rubra*) were evaluated between mid-September and the end of October 2001, after approximately 35 months and 31.5 months incubation, respectively, of the Autumn and Winter applications (table 3). The canopies of all trees treated with Garlon or the Garlon plus Arsenal mixture appeared dead. The canopies of trees which received the FG plus water treatment averaged at least 92 percent dieback and 92 percent transparency, whereas untreated "control" *Q. alba* and *Q. rubra* trees averaged, respectively, no more than 25 and 30 percent dieback and 14 and 26 percent transparency. Of the trees receiving FG plus water, Garlon, and Garlon plus Arsenal, respectively, 35 percent, 8 percent, and 11 percent supported live sprouts at the root crown in September-October 2001.

Lateral Root Survival

Overall, both, one, or neither of the two sampled roots were dead, respectively, on 44 percent, 15 percent, and 41 percent of the study trees at the time of their evaluation. Root mortality was incomplete in all but 3 of the 48 combinations of season, chemical, and tree species. All treatment combinations incubated 28 months or less revealed substantial root survival (tables 1, 2, and 3). All *Q. velutina* roots in the Autumn Garlon treatment at LLCCA were dead after 31.5 months incubation, but 25 percent of corresponding roots at DBCA remained alive after 20.5 months incubation. All *Q. rubra* roots in the Autumn Garlon treatment at DRCA were dead after 35 months incubation, but 20 percent of corresponding roots at DBCA remained alive after 20.5 months incubation. All *Q. rubra* roots in the Winter Garlon treatment at DRCA were dead after 31.5 months incubation, but 70 percent of corresponding roots at DBCA remained alive after 17 months incubation.

Table 1.—*Effects of frill girdle herbicide treatments on lateral root survival, root crown sprouting, and canopy health for* Quercus rubra *and* Q. velutina *at Danial Boone Conservation Area (DBCA)*

Season	Elapsed time[1]	Treatment[2]	Species	DBH		Live sprouts	Live roots	Crown dieback		Foliage transparency	
	months			*cm*	*(SD)*	%	%	%	*(SD)*	%	*(SD)*
Autumn	20.5	Tap Water	*Q. rubra*	20	(5)	90	65	94	(17)	92	(27)
			Q. velutina	22	(4)	50	40	90	(20)	86	(29)
		Garlon 3A	*Q. rubra*	22	(7)	0	20	100	(0)	100	(0)
			Q. velutina	24	(6)	0	25	100	(0)	100	(0)
		G+Arsenal	*Q. rubra*	21	(4)	40	60	100	(0)	100	(0)
			Q. velutina	25	(7)	10	25	100	(0)	100	(0)
Winter	17	Tap Water	*Q. rubra*	22	(10)	80	75	100	(0)	100	(0)
			Q. velutina	24	(3)	50	40	99	(2)	88	(26)
		Garlon 3A	*Q. rubra*	23	(6)	60	70	100	(0)	100	(0)
			Q. velutina	24	(8)	60	60	100	(0)	100	(0)
		G+Arsenal	*Q. rubra*	23	(8)	40	80	100	(0)	100	(0)
			Q. velutina	26	(7)	20	85	100	(0)	100	(0)
Summer	10	Tap Water	*Q. rubra*	21	(6)	90	100	48	(39)	64	(32)
			Q. velutina	27	(9)	50	100	60	(42)	64	(38)
		Garlon 3A	*Q. rubra*	22	(10)	20	80	100	(0)	100	(0)
			Q. velutina	22	(4)	10	100	100	(2)	92	(24)
		G+Arsenal	*Q. rubra*	22	(4)	20	100	100	(0)	100	(0)
			Q. velutina	22	(4)	50	100	100	(0)	100	(0)
Control		None	*Q. rubra*	22	(5)	20	100	17	(8)	12	(4)
			Q. velutina	23	(6)	0	100	22	(11)	18	(5)

[1]Elapsed time: The elapsed time between chemical application and root sampling.
[2]Treatment: All chemicals were applied to edge to edge frill girdles at the rate of 1.0 ml per 5.0 cm tree circumference - Garlon 3A, undiluted; G+Arsenal, half-strength Garlon 3A to which was added 24 ml Arsenal AC per liter of diluted Garlon 3A.

Treatment Factor Effectiveness

As expected, SLR indicated that the elapsed time between treatment and final evaluation (ranging between 10 and 35 months in this study) was directly related to the level of root mortality observed (table 4). Autumn treatment was more effective than Winter or Late Summer treatment. All FG treatments were more effective than no treatment at all. The FG plus water treatment was less effective than either of the herbicide treatments, and the FG plus Garlon treatment was most effective. Root mortality occurrred more slowly in *Q. alba* and *Q. rubra* than in *Q. velutina*.

DISCUSSION AND CONCLUSIONS

Barriers to root graft transmission of the oak wilt pathogen must separate all infected roots (inside the barrier) from all healthy root systems outside the barrier. They are expensive and time-consuming to install. In residential and park settings, the installation and effectiveness of barriers receive close public scrutiny, and the forest health professional's credibility is at stake.

To accomplish an effective barrier, one must recognize that by the time a tree at the epicenter's edge is displaying symptoms, the pathogen has often already reached the root systems of trees to which the symptomatic tree is grafted. Therefore, barriers need to be placed outside non-symptomatic trees which are close enough to symptomatic trees to be grafted directly to them (Bruhn and others 1992). In other words, an immediately effective barrier (for example, one established with a vibratory plow) must be established just beyond the equivalent of one year's disease spread outside the visible epicenter margin.

Proper barrier placement, recognizing that some non-symptomatic trees at the margin of an epicenter are already infected, creates the impression that at least one tier of non-symptomatic trees is being "sacrificed". In the process of modeling the expansion of oak wilt epicenters in northern pin oak stands (*Q. ellipsoidalis* E. J. Hill) growing on sandy soils in Michigan's Upper Peninsula (Bruhn and others 1992), radial expansion of up to 12.5 m (between two trees with combined dbh of 75.7 cm) was documented.

Table 2.—*Effects of frill girdle herbicide treatments on lateral root survival, root crown sprouting, and canopy health for* Quercus alba *and* Q. velutina *at Little Lost Creek Conservation Area (LLCCA)*

Season	Elapsed time[1]	Treatment[2]	Species	DBH		Live sprouts	Live roots	Crown dieback		Foliage transparency	
	months			*cm*	*(SD)*	*%*	*%*	*%*	*(SD)*	*%*	*(SD)*
Autumn	31.5	Tap Water	*Q. alba*	22	(8)	50	80	88	(27)	83	(36)
			Q. velutina	29	(10)	30	5	100	(0)	100	(0)
		Garlon 3A	*Q. alba*	18	(7)	0	15	100	(0)	100	(0)
			Q. velutina	24	(10)	11	0	100	(0)	100	(0)
		G+Arsenal	*Q. alba*	19	(6)	10	45	100	(0)	100	(0)
			Q. velutina	27	(10)	0	20	100	(0)	100	(0)
Winter	28	Tap Water	*Q. alba*	15	(3)	60	60	100	(0)	100	(0)
			Q. velutina	23	(7)	20	15	100	(0)	100	(0)
		Garlon 3A	*Q. alba*	19	(10)	30	40	100	(0)	100	(0)
			Q. velutina	25	(8)	11	30	100	(0)	100	(0)
		G+Arsenal	*Q. alba*	18	(7)	10	50	100	(0)	100	(0)
			Q. velutina	23	(5)	33	35	100	(0)	100	(0)
Summer	21	Tap Water	*Q. alba*	18	(8)	40	80	100	(0)	100	(0)
			Q. velutina	19	(3)	80	65	99	(3)	98	(6)
		Garlon 3A	*Q. alba*	21	(8)	30	60	100	(0)	100	(0)
			Q. velutina	24	(10)	22	55	100	(0)	100	(0)
		G+Arsenal	*Q. alba*	18	(9)	0	55	100	(0)	100	(0)
			Q. velutina	25	(9)	0	35	100	(0)	100	(0)
Control		None	*Q. alba*	19	(6)	10	100	18	(11)	17	(8)
			Q. velutina	21	(5)	10	100	28	(27)	26	(27)

[1]Elapsed time: The elapsed time between chemical application and root sampling.
[2]Treatment: All chemicals were applied to edge to edge frill girdles at the rate of 1.0 ml per 5.0 cm tree circumference - Garlon 3A, undiluted; G+Arsenal, half-strength Garlon 3A to which was added 24 ml Arsenal AC per liter of diluted Garlon 3A.

Table 3.—*Effects of frill girdle herbicide treatments on lateral root survival, root crown sprouting, and canopy health for* Quercus alba *and* Q. rubra *at Deer Ridge Conservation Area (DRCA)*

Season	Elapsed time[1]	Treatment[2]	Species	DBH		Live sprouts	Live roots	Crown dieback		Foliage transparency	
	months			*cm*	*(SD)*	*%*	*%*	*%*	*(SD)*	*%*	*(SD)*
Autumn	35	Tap Water	*Q. alba*	20	(3)	50	30	100	(0)	100	(0)
			Q. rubra	26	(13)	40	30	100	(0)	100	(0)
		Garlon 3A	*Q. alba*	19	(6)	10	15	100	(0)	100	(0)
			Q. rubra	25	(8)	11	0	100	(0)	100	(0)
		G+Arsenal	*Q. alba*	22	(4)	10	10	100	(0)	100	(0)
			Q. rubra	22	(6)	20	15	100	(0)	100	(0)
Winter	31.5	Tap Water	*Q. alba*	27	(8)	50	95	92	(18)	92	(24)
			Q. rubra	21	(6)	70	60	94	(19)	92	(25)
		Garlon 3A	*Q. alba*	27	(10)	10	40	100	(0)	100	(0)
			Q. rubra	22	(13)	0	0	100	(0)	100	(0)
		G+Arsenal	*Q. alba*	23	(9)	10	30	100	(0)	100	(0)
			Q. rubra	19	(8)	10	10	100	(0)	100	(0)
Control		None	*Q. alba*	28	(12)	0	100	25	(11)	13	(3)
			Q. rubra	22	(13)	0	95	30	(28)	26	(28)

[1]Elapsed time: The elapsed time between chemical application and root sampling.
[2]Treatment: All chemicals were applied to edge to edge frill girdles at the rate of 1.0 ml per 5.0 cm tree circumference - Garlon 3A, undiluted; G+Arsenal, half-strength Garlon 3A to which was added 24 ml Arsenal AC per liter of diluted Garlon 3A.

Table 4.—*Results of stepwise logistic regression to identify factors associated with root mortality*

EXPLANATORY VARIABLES				PREDICTIVE EVALUATION[3]		
Intercept	Name[1]	Parameter estimate	Pr > Wald Chi-square[2]	Percent Concordant	Percent Discordant	No. pairs
- 3.3682			0.0001	87.1	12.1	369413
	Elapsed Time	0.1533	0.0001			
	Autumn	0.6079	0.0003			
	FG + Water	-0.5612	0.0029			
	FG + Garlon 3	0.5750	0.0025			
	Control	-5.1045	0.0001			
	Q. alba	-1.3066	0.0001			
	Q. rubra	-0.4547	0.0181			

[1]Variable: Elapsed time: The elapsed time between chemical application and root sampling; FG, edge to edge frill girdle treatment; Control, nontreated.
[2]The significance test used by PROC LOGISTIC is based on the Wald statistic (Sokal and Rohlf 1995).
[3]No. Pairs represents the product (Observed) * (Total - Observed), the number of possible pairwise combinations of sampled roots in which one sampled root was dead and the other remained alive, respectively. For each sampled root, SLR assigned a probability, p, that the sampled root would be dead. 'Percent Concordant' is the percentage of pairs for which the larger value of p was associated with the dead root. 'Percent Discordant' is the percentage of pairs for which the larger value of p was associated with the surviving root.

One should anticipate annual expansion over greater distances around epicenters in stands of larger trees, and over shorter distances in stands growing on heavier soils (Bruhn and others 1992). Placement of a barrier which is not immediately effective (for example, a band of herbicide-treated trees) should take into account the epicenter expansion anticipated during the time required for the barrier to become effective. This would result in the sacrifice of additional land area and the healthy trees growing there.

A variety of herbicides has been used operationally in efforts to create barriers to root graft spread of oak wilt. Frill girdle application with Garlon 3A was used rather than basal stem spray application of Garlon 4 to achieve greater effectiveness on large diameter stems. The Specimen Label for Dow AgroSciences' Forestry Garlon 4 recommends use of this product as a basal bark treatment on stems less than 6 inches in basal diameter.

Frills were created with hacks placed edge-to-edge rather than the more standard spacing of 3 inches between hacks (Miller 1991), to accomplish the most rapid and complete treatment effect. Though extremely laborious, frills cut at a downward angle into the wood have the advantage of trapping the applied herbicide, whereas an unacceptable portion of herbicide applied to a chainsaw frill might be washed out by rain into the soil.

Tordon RTU is often suggested as a good candidate for causing rapid root death. However, Bruhn and Haugen (unpublished) have each observed that Tordon RTU applied via cut frill to create barrier zones of trees in late summer failed to halt the radial expansion of oak wilt epicenters in the Upper Peninsula of Michigan and in northeast Iowa, respectively. In each case, the disease expanded beyond the barrier of treated trees in a single year. In addition, it was observed that the application of chainsaw frills can result in the creation of hazard trees, as the structural integrity of the treated tree is compromised.

In a small trial in Iowa (Walkowiak, Bruhn and Haugen, unpublished), Tordon treated trees sprouted and Garlon 4 treated trees did not, suggesting that a trichlopyr herbicide might be more effective than Tordon RTU in killing the root crown. In the larger study described in this paper, the canopies of trees treated with Garlon 3A or the Garlon 3A plus Arsenal mixture all appeared dead 35 months after late Autumn treatment; however, some of the trees of each species still supported live sprouts at the root crown. Canopy mortality occurred well in advance of root mortality, and absence of sprouts at the root crown is not a dependable indication of root system mortality (tables 1, 2 and 3).

Our study is the first to evaluate the timing of tree root mortality following herbicide treatments in forestry applications. The failure of

herbicides to cause rapid root death is disappointing. If herbicides require 3 years to reliably render the root systems of treated trees disfunctional for root graft transmission of the pathogen, then the herbicide barrier needs to be placed approximately 4 years' radial spread beyond the currently visible epicenter margin. This could result, for example, in the "sacrifice" of a belt of non-infected (as well as non-symptomatic) trees 50 m deep rather than 12.5 m deep as might be the case for a barrier zone constructed with a vibratory plow. In most cases, a herbicide barrier with this type of location requirement would be considered too inefficient.

We have tested two highly recommended herbicide treatments using a vital staining technique which can easily be used to evaluate other candidate herbicides. No herbicide treatment has yet been demonstrated to provide root death quickly enough to ensure the discontinuity of root systems necessary to reliably contain oak wilt disease epicenters. However, the extent of root system mortality required to effectively block pathogen transmission is not known. Considering the costs involved, establishment of barriers at a distance sufficient to permit herbicides to take adequate effect seems currently unacceptable.

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THE NEED FOR SILVICULTURAL PRACTICES AND COLLECTION OF BUTTERNUT GERMPLASM FOR SPECIES CONSERVATION

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ABSTRACT.—Butternut is a short-lived tree and is declining in numbers for a variety of reasons ranging from changing land use to aging forest stands, seed predation and lack of suitable conditions for reproduction. However, the major reason for the dramatic decrease in butternut populations throughout its range in North America is the lethal canker disease caused by the fungus *Sirococcus clavigignenti-juglandacearum* that is believed to be an exotic pathogen. The fungus has killed up to 80 percent of the trees in some states and is threatening butternut's survival as a viable species in North America.

There is a critical need for additional collections of butternut (*Juglans cinerea* L.) from throughout its range in the eastern United States and the development of silvicultural practices targeted at butternut regeneration. Butternut is being killed throughout its range in North America by the fungus *Sirococcus clavigignenti-juglandacearum*. Diseased trees are eventually overwhelmed by multiple branch and stem cankers. First observed in 1967 in southwestern Wisconsin, the disease is now affecting over 90 percent of the living trees and has killed up to 80 percent of the trees in some states and is threatening butternut's survival as a viable species (Ostry 1998). Eastern black walnut (*J. nigra* L.) and heartnut (*J. ailantifolia* var. *cordiformis* Carr.) are other natural hosts but not severely damaged (Ostry and Pijut 2000). The potential host range among *Juglans* species is cause for concern outside of North America as well (Nair 1999).

The fungus was described as a new species in 1979 and evidence thus far strongly suggests that this pathogen is an exotic from an unknown origin outside of North America (Furnier and others 1999). The fungus can become airborne and dispersed long distances during rain (Tisserat and Kuntz 1983) and can be seedborne on butternut and eastern black walnut (Innes 1998). Insects (Bergdahl and Halik 1998, Katovich and Ostry 1998) and perhaps birds are also probably involved in the long distance spread of the fungus explaining how the pathogen has been able to move rather quickly throughout the widely scattered populations of butternut in the United States and Canada.

Butternut is a small- to medium-size tree seldom exceeding 75 years of age and is shade intolerant. It commonly grows on rich loamy soils as well as on drier rocky soils of limestone origin (Rink 1990). While not occurring in pure stands, butternut can be locally abundant in mixed hardwood forests and often grows as a riparian species. Butternut has never been a major commercially important species, however, it is valued for its wood for furniture, paneling, specialty products, and carving, its flavorful nuts, wildlife mast, and for its contribution to forest diversity. Its value is increasing as mature trees are becoming increasingly scarce and the disease, if present, is killing regeneration.

KEY DATES IN THE BUTTERNUT CANKER HISTORY

To our knowledge, there are no written records or evidence based on many dissections and aging of cankers on diseased trees in the field that indicates the presence of the disease in the Lake States prior to the early 1960's, however, evidence from trees in the southern portion of its range suggests that the disease has been present in North America earlier than the 1960s. Rapid spread of the pathogen was documented in Wisconsin using permanent plots

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that indicated that from its original presence in western Wisconsin in 1976 the fungus had moved throughout the range of butternut in the state by 1992.

It was not until 1979 that the fungus was proved to be a new species never before recorded on any host species (Nair and others 1979) and efforts to determine its origin have not been successful. In Canada, the disease was first reported from Quebec in 1990, Ontario in 1992, and from New Brunswick in 1998. In 1992 Minnesota enacted a moratorium on the harvest of healthy butternut on state lands. This was followed by harvest restrictions on healthy butternut on federal lands by the USDA Forest Service in 1993. Butternut is a Regional Forester Sensitive Species in the Eastern Region on 13 of the 16 National Forests and is listed as a sensitive species or a species of concern in most states in which butternut is a common component of hardwood forests. Management guidelines have been put in place to conserve butternut in several of those states.

THE NEED FOR CONSERVATION STRATEGIES

Conservation assessments for butternut are being prepared in the United States and Canada. Criteria for determining the need for conservation strategies for species at risk include the following: the species is naturally rare; there is an uncertain viable seed source; its range is decreasing; its habitat is in demand for other uses; poor regeneration after usual forest practices; loss due to hybridization; demand for species for special purposes; and there is a serious threat from disease (Loo 1998). Butternut meets all these criteria.

In order to improve prospects for the recovery of butternut there are several key issues that must be addressed. First, increased cooperation among researchers, landowners, and resource managers is needed to identify and retain selected healthy butternut wherever they occur. Second, we must increase the collection of scionwood from healthy trees that may have disease resistance to preserve genetic diversity within butternut across its range while it is still possible. Third, develop a greater understanding of the specialized habitat and site requirements of butternut so that we can favor natural and artificial regeneration. Fourth, develop practical butternut management prescriptions that can be implemented by landowners and managers.

Both *ex situ* (conservation of planted seedlings or in clonal archives, seed banks, etc.) and *in situ* (conservation within protected areas of natural occurrence) conservation strategies are needed and warranted for butternut (Mcllwrick and others 2000). Each strategy has its advantages and disadvantages. Several efforts to conserve butternut are underway in the United States and Canada. Brief summaries of two examples from the north central region representing *in situ* and *ex situ* projects follow.

CLONAL ARCHIVES

High mortality of butternut in all age classes and lack of adequate regeneration, both resulting from multiple stress agents, aging of the resource, changing land uses, and butternut canker all contribute to the need to conserve individual trees exhibiting superior qualities among populations of butternut outside (*ex situ*) of the natural stands where they are found.

Over the years, very few selected butternut cultivars were described and only a few have survived and are available for propagation on a limited basis. All of these trees were selected for their superior nut qualities but they do not represent the geographical range or diversity of the species and their resistance to butternut canker is unknown.

Since 1992 we have established five butternut clonal archives located in Minnesota, Wisconsin, Illinois, New York, and Vermont consisting of trees selected from 14 states on the basis of their phenotypic resistance to butternut canker in the field. The scionwood, collected from these trees that exhibit disease resistance were grafted onto black walnut rootstock and the grafted trees were established in replicated plantings. These trees will be screened for their level of disease resistance and genetic typing will be used to examine genetic variation among the trees and eventually the patterns of inheritance of disease resistance. In spite of high disease incidence among the surrounding trees, many of the trees we have collected scionwood from have remained disease-free for over 10 years, providing encouraging evidence that disease resistance does exist in butternut populations.

The shortcoming of this effort thus far is that there has been little investigation of the genetics of butternut. Lacking is an adequate study of the genetic structure and diversity of butternut across its range. This information is critical to designing an effective conservation strategy and is needed to guide us in determining what germplasm, how many genotypes, and where in its range we put our collection emphasis. Only a limited amount of resistance screening has been carried out owing to a lack of an efficient

screening technique so we do not know the level of resistance or the mechanisms of resistance of the phenotypes collected thus far.

We have detected two different bark phenotypes among the butternut we have studied: a light gray, shallow fissured type most often associated with cankered trees and a dark gray, deeply fissured type often associated with healthy trees. In addition, there are intermediate types. In one 40-acre woodlot of the 544 butternut examined, 92 were disease-free, 67 (73 percent) of them being the dark-deep bark phenotype. Of the 452 diseased or killed butternut, 93 (21 percent) were the dark-deep phenotype and 310 (69 percent) were the light-shallow phenotype. Our research goal is to determine if this bark trait may be valuable in identifying potentially resistant trees and if it may be an identifiable heritable trait that could be used in selection and breeding.

REGENERATION STUDY

Conservation of individuals or populations of butternut within its natural habitat (*in situ*) is preferable to removing individual genotypes to collections located distant to their origins, however, this is not always feasible if trees cannot be protected. Throughout its range individual healthy butternut occasionally can be found growing alongside severely diseased trees. Silviculture, intended or not, can affect genetic change in a species. Guidelines to retain trees that may have disease resistance have been proposed (Ostry and others 1994). No specific silvicultural practices have been developed for butternut; however, butternut is closely related to black walnut so many management recommendations are similar for both species.

In 1993, on the Nicolet National Forest in Wisconsin, we established what we believe to be the only large-scale study designed to conserve and regenerate butternut according to the suggested guidelines. Our objectives were to determine the validity of the tree retention guidelines and to develop stand conditions to successfully regenerate butternut naturally and by planting.

A pending 160-acre sale unit in a northern hardwood stand on the Laona Ranger District containing 15 square feet basal area (BA) of butternut presented us with a unique opportunity to apply the guidelines. The 1938 air photos show numerous roads and grades through a moderately stocked hardwood stand, perhaps 20 to 30 years old. It seems that this area was never plowed. The site is habitat type AH (*Acer/Hydrophyllum*), a rich mesic type.

In the 1970s to the early 1980s, there were two timber sales in the area. Records are not complete, but these sales typically involved heavy American elm (*Ulmus americana*) and trembling aspen (*Populus tremuloides*) cutting. In 1994 through 1997 the Old Town Road sale resulted in the current administrative study.

In June 1993 marking crews were trained in the identification of butternut canker and in applying the marking guide. The marking guide was designed to increase seed production and to conserve any butternut that may have disease resistance and those trees that were likely to survive the 15 years between stand entries.

All trees with more than 70 percent live crown and less than 20 percent of the combined circumference of the bole and root flares affected by cankers were retained. In addition, all trees with at least 50 percent live crown and no cankers on the bole or root flares were retained. Dead butternut and trees of poor vigor were marked for harvest. A total of 1,165 butternut in the stand was marked; 43 percent reserved and 57 percent cut. Various pre-harvest habitat and plant data were collected. Several large, healthy butternut were marked for scionwood collection to be included in the clonal archives.

The sale unit was divided into three equally-sized blocks each sub-divided into the following treatments: one 5 acre clearcut; two 1 acre clearcuts; eight 2 acre blocks with two cut to 60 square feet BA, two cut to 30 square feet BA, two clearcuts, and two no-cut controls. Selection cut buffers separated all treatment blocks. One block was to be direct seeded, one to be planted, and one to be a natural regeneration control. The stand was harvested according to the prescription from 1994 to 1997. Direct seeding was done in the fall of 1995. Butternut seedlings, 2-0 stock from the same seed crop as the direct seeding, were planted in tree shelters in 1998. The pattern for both the direct seeding and the planting was to place a row in each of the cardinal directions from the center of the plot. Regeneration surveys were made in 1999 and 2001.

Early observations have revealed that recruitment of new butternut regeneration continued from 1999 to 2001 but approximately 60 percent of the seedlings are infected and the regeneration is not uniform throughout the site. Stocking from direct seeding and planting 2-0 stock both appear unsatisfactory. Hardwood sprouts and heavy shrub growth are outcompeting much of the butternut regeneration.

Mortality of planted seedlings has been high but is not known to be associated with cankers. Not unexpectedly, few of the infected stumps have sprouted and the majority of those sprouts are infected. Canker-related mortality of reserve trees continues but trees in the intermediate cut blocks, in the no-cut controls and in the buffers look to be more vigorous than those in the clearcut blocks. Previously tagged healthy trees, those selected for the clonal archive, remain healthy.

Butternut is widely recognized as shade intolerant. As expected, the more severe cutting treatments produced a more favorable seedling environment, but not with the consistency one would hope for. At this time, the reserve trees that are serving as the seed source seem to be in better health in the moderate thinning treatments, and the seedlings are growing more vigorously in the heavier thinning or clearcut treatments. Therefore, we speculate that there should be two prescriptions for stands where butternut is of concern, one to release the mature trees and promote seed production, and one that would provide optimum conditions for the seedlings in the new stand.

Wisconsin's pre-harvest white-tailed deer herd in the year 2000 was estimated at over 1.7 million. That year's harvest of over 618,000 established an all time record for any state. Even at that, the 2001 herd was estimated at more than 1.6 million. The butternut study area is in a deer management unit that has been fairly stable and consistently closer to goal than many others in the Northern Forest. However, deer have impacted the butternut regeneration in two ways, by browsing and antler rubbing. Unless it is suppressed by shade or affected by cankers or repeated browsing, butternut will quickly grow out of the browse zone. Annual height growth over 3 feet is common. While the frequency of browsed seedlings can be high, the majority of them are not seriously damaged.

Deer browse may be a net asset by controlling the competing trees and brush. Deer tend to use salient saplings to rub the velvet off their antlers as they harden. This behavior also has some role in mating and territorial dominance. Butternut tends to occupy open areas in stands where deer activity is high and may account for the high incidence of damage observed, however, based on observations deer may preferentially select butternut seedlings for unknown reasons. This damage may be as significant as that caused by browsing.

The most frequently observed seed dispersing animals are red squirrels and chipmunks. Grey squirrels are present, but not in the numbers that they might be found in nearby oak (*Quercus* L.) dominated stands.

Bitternut hickory (*Carya cordiformis* (Wangenh.) K. Koch) is an indicator species for the forest type in this study area, and is present within the stand. The stand developing in the clearcut blocks includes a considerable number of seed origin basswood as well as stump sprouts. There is an interesting contrast in the stump sprouts. In the clearcuts, the sprouts are able to grow out of browse height, while in the lighter cuts they are suppressed and heavily browsed with corresponding high levels of sprout mortality. White ash (*Fraxinus americana* L.) and hard maple (*Acer saccharum* Marsh.) will continue as the dominant species, ash being favored in the more open treatments, maple in the more closed canopy. Gone from the clearcut blocks are the elm and yellow birch (*Betula alleghaniensis* Britton).

FUTURE OUTLOOK

Although butternut is an excellent candidate for the application of gene conservation and restoration techniques, we are lacking the knowledge in many critical areas in the biology of both the host and pathogen to efficiently and successfully employ them at this time. However, there is still justification to immediately begin to build the framework of a restoration program and initiate collections of potentially valuable germplasm in the wake of the rapid decline in the resource.

Butternut canker is another example of a disease that is currently having negative impacts on forest ecosystems, and perhaps other impacts that may not be obvious for years to come. Considering not only butternut canker, but also several other diseases that are affecting our hardwood forests such as oak wilt, oak decline, dogwood anthracnose, Dutch elm disease, and a number of insect pests and environmental stresses, it is obvious that we have to address multiple issues in the management of our hardwood forests.

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LEGUME GROUND COVERS ALTER DEFOLIATION RESPONSE OF BLACK WALNUT SAPLINGS TO DROUGHT AND ANTHRACNOSE

J.W. Van Sambeek[1]

ABSTACT.—Growth and premature defoliation of black walnut saplings underplanted 5 or 6 years earlier with six different ground covers were quantified in response to a summer drought or anthracnose. Walnut saplings growing with ground covers of hairy vetch, crownvetch, and to a lesser extent sericea lespedeza continued to have more rapid height and diameter growth than saplings in resident vegetation. Walnut saplings in a dense cover of sericea lespedeza had more rapid defoliation during a summer drought than saplings growing in a ground cover of resident vegetation. During the following year with nearly normal precipitation, walnut saplings growing in ground covers of resident vegetation had more rapid defoliation than walnut saplings underplanted with hairy vetch, crownvetch, and sericea lespedeza. Walnut saplings that had been underplanted with annual legumes that had failed to reseed themselves (crimson clover or Korean lespedeza) had defoliation rates similar to walnut saplings in resident vegetation. Regression analysis with a full first and second-order polynomial indicated that approximately 50 to 70 percent of the variation in the subsequent annual growth increment was a function of premature defoliation and sapling stem diameter, trunk cross-sectional area, or stem volume.

Walnut anthracnose caused by the fungus *Gnomonia leptostyla* (Fr.) Ces. & de Not., is the most significant pathogen found in most black walnut (*Juglans nigra* L.) plantations (Kessler 1984, 1988). Berry (1960) reported that premature defoliation caused by anthracnose can result in reduced tree growth and development of poorly filled, low-quality nuts. Primary infections initiated from ascospores in May under favorable weather produce conidia that produce secondary infections. These infections can multiply rapidly and produce significant defoliation as early as July or August (Black and others 1977, Black and Neely 1978). Todhunter and Beineke (1984) found that the severity of mid-September defoliation, presumably in response to anthracnose, was correlated with annual stem diameter growth but not annual increment in stem height or stem volume. Conversely, Funk and others (1981) found that selection of seed sources with some resistance to anthracnose resulted in a 7 percent increase in height growth.

Neely (1979, 1986) suggested that the incidence or severity of anthracnose can be altered through site modification and increase in foliage nitrogen content. Kessler (1985) found that an overwinter cover of autumn olive leaves reduced the number of ascospores, the primary inoculum, discharged from infected fallen walnut leaves. In addition, nitrogen-fixed and released from autumn olive increases the walnut foliage nitrogen content and reduces walnut susceptibility to secondary infections initiated by conidia released from the primary infections. Use of nitrogen fertilizers can also reduce the severity of anthracnose (Neely 1981).

Underplanting herbaceous legumes in walnut plantations could potentially reduce the severity of anthracnose either in response to increased soil nitrogen or by disrupting ascospore dispersal. Underplanting walnut saplings with annual and perennial legumes has been shown to increase foliage nitrogen content (Van Sambeek

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and others 1986). Underplanting herbaceous legumes that defoliate later in the growing season than walnut could also cover and accelerate overwinter decomposition of infected walnut leaves within the litter layer thereby reducing ascospore dispersal in the spring. In contrast, underplanting legumes such as the vetches, which initiate rapid vegetative growth in the spring, would cover the developing perithecia on fallen infected leaves and physically disrupt ascospore dispersal.

Quantifying the incidence and severity of leaf-spotting pathogens can be very time-consuming if the actual numbers of lesions are counted. In addition, premature defoliation of severily-infected leaflets early in the growing season can underestimate abscission rate and severity (Kessler 1984). Crown estimates of defoliation near the end of the growing season are also unreliable because black walnut can initiate a second flush following severe premature defoliation or under favorable environmental conditions continue new shoot growth late into the summer masking defoliation of older leaves (Reid, personal communication). Theoretically, quantifying leaflet abscission through the growing season could be a reliable alternative to making lesion counts or estimates of crown defoliation. The objective of my study was to quantify leaflet abscission through the growing season and to determine the impact of premature defoliation on the growth rate of black walnut saplings grown with six different ground covers.

MATERIALS AND METHODS

The study was initiated in a 5-year-old walnut plantation affected by anthracnose during the previous summer. The plantation was on an upland ridge in Jackson County, IL (37º 34′ latitude north, 89º 16′ longitude west) on soils of the Alford silt loam series (fine-silty, mixed, mesic, Typic Hapludalfs). The planting was established using a randomized complete block design with three blocks and factorial combinations of six ground covers, with and without spot weed control, and co- or post-establishment of walnut seedlings with ground covers. Each block consisted of twenty-four 12 x 21 m plots that were planted with 24 or 28 walnut seedlings on 3 x 3 m spacing.

The following six ground covers were seeded the preceding fall and overseeded in the spring:

1) control consisting of old-field resident vegetation,
2) seeding with hairy vetch (*Vicia villosa* Roth),
3) crownvetch (*Coronilla varia* L.),
4) sericea lespedeza (*Lespedeza cuneata* (Dum. Cours.) G. Don),
5) Korean lespedeza (*L. stipulacea* Maxim.), and
6) a mixture of Korean lespedeza and crimson clover (*Trifolium incarnatum* L.).

Korean lespedeza did not become established under the heavy stand of crimson clover, thus this treatment was essentially crimson clover. Half of the walnut seedlings were planted with the cover crop and the other half the following spring after ground covers were established.

Weed control around each walnut seedling consisted of spraying 6.4 liters per hectare glyphosate and 4.4 kilograms per hectare simazine in 1.5-m diameter spots each spring for 3 years in one half of the plots for each ground cover. Whether walnut saplings were established with or after the ground cover and with or without chemical weed control had no effect on defoliation rate during the fifth and sixth growing season, so these factors and their interaction terms were pooled into the analysis of variance error term. Percentage plot coverage as either live legume or as bare ground and dead vegetation was visually estimated in the spring, summer, and fall, and averaged to estimate percentage desired ground cover. Rainfall and temperature data are averages from NOAA reporting stations at Carbondale, IL (13 km north of the planting) and Anna, IL (19 km south of the planting).

Eight trees near the center of each plot were selected for evaluating defoliation in year 5 and year 6. A fully expanded leaf was selected midway along the new growth of the terminal shoot and four mid-crown branches (one in each cardinal direction) and marked with plastic flagging both years. Mid-June was selected for selecting and marking because most shoots were setting terminal buds and little if any defoliation from secondary infections had occurred.

A leaf midway along the new shoot was used because the number of expanding leaflets could not be counted on the more apical leaves. The more basal leaves were not used because they are the sites for both primary and secondary anthracnose infections. The total number of leaflets on all five marked leaves on each tree was recounted in early August and mid-September of year 5 and in mid-July, mid-August, and mid-September of year 6. Defoliation was determined on an individual tree basis as the percentage of abscised leaflets compared to the original number of leaflets recorded on the five marked leaves.

Marked trees were measured annually in centimeters for stem height (h) and stem basal diameter at ankle height (d.a.h.) and used to calculate trunk cross-sectional area as 0.78 $(d.a.h.)^2$ and stem volume as $(d.b.h.)^2h$ for each marked tree. For each plot, mean defoliation was determined, tested for normal distributions, subjected to analysis of variance, and then means were separated using the Duncan's new multiple range test when significant differences existed at the $p < 0.05$ percent level. Correlation between defoliation and annual growth increment of current and subsequent year were done using either individual tree data or plot means. Because both ground cover treatments and defoliation affected walnut growth, size variables and mid-season defoliation were fitted to a full first- and second-order polynomial followed by analysis of variance on coefficients to determine which terms had a significant effect on the following year's growth.

PERSISTENCE OF GROUND COVERS

During the fifth growing season, crownvetch, a cool-season perennial with decumbent stems, had the highest percentage of the desired ground cover (bare or mulch-covered soil and live legume vegetation) when averaged across the spring, summer, and fall survey (table 1). Sericea lespedeza, a warm-season perennial, also had a higher percentage of the desired ground cover than the other four ground cover treatments.

Although both ground covers had effectively suppressed the resident vegetation, they had very different growth phenologies. The crownvetch ground cover had actively growing vegetation and flowered during the spring inventory with mostly dormant vegetation during the summer and fall inventories. In contrast, the sericea lespedeza ground cover had actively growing vegetation during the summer and was flowering during the fall inventories. All treatments were bush-hogged each fall after a killing frost to reduce risk of wildfires; this treatment also effectively mulched most plots with sufficient dead vegetation to partially suppress resident vegetation the following spring.

Although hairy vetch, a cool-season annual vine, produced one of the most dense covers during establishment of the walnut seedlings (Van Sambeek and Rietveld 1982), only isolated dense patches of hairy vetch were present each spring during the fifth, sixth, and seventh growing season. These dense patches of hairy vetch usually died before producing mature seed while relatively thin stands produced sufficient seed to initiate isolated dense stands the following year. Unlike other ground cover treatments, cheat (*Bromus secalinus* L.) dominated the resident vegetation in plots originally seeded with hairy vetch. This grass germinated in the fall, flowered in late spring, and mulched these plots through the growing season.

The desired vegetative cover for crimson clover, Korean lespedeza, and resident vegetation averaged approximately 20 percent primarily because of the mulch of dead resident vegetation produced by fall mowing (table 1). In year 5 to year 7, resident vegetation was predominately a mix of broomsedge (*Andropogon virginicum* L.) and composites that consisted mostly of *Solidago, Aster*, and *Erigeron* species. Crimson clover, an upright cool-season annual, had produced a dense stand the spring of year 1;

Table 1.—*Desired fourth-year ground cover (sum of exposed soil, dead vegetation, and seeded legume) and annual growth increment for height, basal diameter, trunk cross-sectional area, and stem volume for walnut saplings 5 to 7 years after planting with six ground covers*

Ground cover	Desired vegetative cover	Year 4		Year 5 to 7 annual growth increment			
		Height	Basal diameter	Height	Basal diameter	Trunk area	Stem volume
	— % —	— cm —	— cm —	—cm—	—cm—	—cm^2—	—cm^3—
Crownvetch	91 a	157 b	3.5 b	70 b	1.34 a	13.5 b	6120 b
S. lespedeza	64 b	169 b	3.0 bcd	53 c	1.02 b	8.9 c	3680 c
Hairy vetch	27 c	219 a	4.6 a	85 a	1.61 a	18.2 a	9650 a
Crimson clover	19 c	157 b	3.4 bc	50 c	0.92 b	8.5 c	3430 c
K. lespedeza	25 c	123 c	2.6 d	45 c	0.75 b	6.0 c	2280 c
R. vegetation	18 c	147 bc	2.8 cd	45 c	0.80 b	6.8 c	2650 c

Means followed by the same letter within columns are not significantly different at the 0.05 level according to Duncan's new multiple range test.

however, either the new seed failed to germinate or seedlings failed to overwinter the second year. Korean lespedeza, a low-growing warm-season annual, did not effectively suppress resident vegetation in year 1 and was no longer found in year 5. In general, the desired vegetative cover and species remained relatively stable until the walnut crown competition factor reached 80 to 100 and resident vegetation gradually changed to the annual cheat grass that first dominated areas seeded to hairy vetch.

LEGUMES AND WALNUT GROWTH

After four growing seasons, walnut seedlings established with a hairy vetch ground cover were taller than walnut saplings with the other five ground covers (table 1). There was a trend for walnut saplings established with Korean lespedeza and resident vegetation to be shorter than for saplings established with ground covers of crownvetch, crimson clover, and sericea lespedeza. Stem basal diameter confirmed this trend, where walnut saplings established with hairy vetch, crownvetch, crimson clover, and sericea lespedeza were larger in diameter than walnut saplings established with a ground cover of Korean lespedeza or resident vegetation.

During the fifth through seventh growing season, walnut saplings established with a ground cover of hairy vetch continued to have the largest annual height and diameter growth increments although hairy vetch was a minor component of the ground cover (table 1). Saplings growing in dense ground covers of crownvetch also had larger annual height and diameter increments than did saplings growing in a relatively dense ground cover of sericea lespedeza. When averaged across the fifth through seventh growing season, walnut saplings in the dense ground covers of sericea lespedeza had annual height and diameter growth increments similar to those established with crimson clover, Korean lespedeza, or resident vegetation. There was a trend for annual increment of walnut established with crimson clover to gradually decline from the fifth to the seventh growing season (9.0, 7.6, and 10.7 mm, respectively) and annual increment for walnut established with sericea lespedeza to gradually increase (7.9, 8.8, and 13.3 mm, respectively) while annual growth increments remained relatively constant for walnut established with Korean lespedeza or resident vegetation (7.1, 6.2, and 9.7 mm, respectively).

LEGUMES AND WALNUT DEFOLIATION

Leaflet abscission on tagged leaves occurred later during the growing season in year 5 than it did in year 6 (table 2). In year 5, 25 percent of the tagged leaflets had abscised in mid-August and premature defoliation gradually increased to near 50 percent by mid-September. Visual examination of the tagged leaflets in year 5 revealed most leaflets had few brown lesions characteristic of anthracnose. Records from the National Oceanographic and Atmospheric Sciences for two nearby weather stations indicate in year 5 the planting probably had above normal precipitation in April (29.5 cm), May (23.7 cm), and June (16.7 cm) followed by below normal precipitation in July (2.5 cm), August (3.2 cm), and September (2.4 cm). In addition, there were more than 40 days with precipitation in the spring followed by only 11 days with precipitation during the summer. The below normal late summer precipitation, the few periods of 12 or more hours of leaf wetness, lack of secondary anthracnose infections, and delayed leaflet abscission indicate premature defoliation was in response to moisture stress induced by the summer drought.

Table 2.—*Mean number of leaflets per leaf and percentage defoliation of tagged leaflets on walnut saplings grown in six different ground covers*

Ground cover	Leaflets per leaf	YEAR 5 DEFOLIATION Early August	YEAR 5 DEFOLIATION Middle Sept.	YEAR 6 DEFOLIATION Middle July	YEAR 6 DEFOLIATION Middle August	YEAR 6 DEFOLIATION Middle Sept.
	—no.—	— % —	— % —	— % —	—%—	—%—
Crownvetch	17.1 ab	14 a	47 a	17 a	38 a	84 a
S. lespedeza	16.4 b	19 b	62 b	16 a	37 a	82 a
Hairy vetch	17.8 a	14 a	44 a	14 a	43 a	86 ab
Crimson clover	17.1 ab	14 a	43 a	15 a	38 a	92 c
K. lespedeza	16.3 b	14 a	50 a	16 a	38 a	91 bc
R. vegetation	16.2 b	15 a	47 a	17 a	43 a	93 c

Means followed by the same letter within columns are not significantly different at the 0.05 level according to Duncan's new multiple range test.

In contrast to year 5, almost 40 percent of the tagged leaflets had abscised before mid-August and rapidly increased to more than 80 percent by mid-September of year 6 (table 2). Many of the tagged leaflets had large brown lesions characteristic of secondary anthracnose infections. Climatological data for year 6 suggest slightly below normal precipitation during April (9.3 cm), May (11.6 cm), and June (4.6 cm) followed by near normal precipitation during July (7.4 cm), August (8.4 cm), and September (15.7 cm). Of greater significance may be the number of days with precipitation because spore germination and hyphen infection require extended periods of leaf wetness. In year 6, there were 35 days with precipitation in the spring followed by more than 25 days with precipitation during the summer supporting the observation that premature defoliation found in year 6 was in response to secondary anthracnose infections.

In early August of the fifth growing season, defoliation of tagged leaflets of walnut trees with a ground cover of sericea lespedeza was greater than that of walnut with any of the other ground covers (table 2). Accelerated defoliation of walnut growing with sericea lespedeza during a summer drought was even more pronounced by mid-September compared to walnut saplings growing in a nearly dormant cover of crown vetch and mostly non-existent cover of hairy vetch, crimson clover, and Korean lespedeza. Sericea lespedeza is a deep-rooted perennial legume with rapid vegetative growth during the summer and may have depleted available soil moisture more rapidly than resident vegetation or the cool-season legume ground covers.

In mid-July and mid-August of the sixth growing season, defoliation of tagged leaflets averaged 15 and 40 percent, respectively, with no statistical differences for walnut trees growing with any of the six ground covers (table 2). Near the end of the growing season, walnuts growing in ground cover treatments dominated by resident vegetation had abscised more than 90 percent of the tagged leaflets compared to only 84 percent of the tagged leaflets on walnut saplings growing with hairy vetch, crownvetch, or sericea lespedeza. The latter three ground cover treatments had the highest percentage of legume ground cover during the growing season. Hypothetically, these ground covers made more soil nitrate nitrogen available to the walnut saplings reducing intensity of anthracnose infections.

DEFOLIATION AND WALNUT GROWTH

Several studies have shown that most of the annual height growth for walnut saplings is complete by late-July in southern Illinois or Missouri (Bey and others 1971, Lucier and Hinckley 1982, Van Sambeek and others 1989). These studies also show that diameter growth occurs over a slightly longer time interval and is frequently terminated by late August. Theoretically, premature defoliation occurring in August or September should minimally affect growth during the same year.

Todhunter and Beineke (1984) found that mid-September defoliation in response to anthracnose was not correlated with height growth of walnut saplings during the same year but was negatively correlated with diameter growth. Evidence from other fungal diseases suggests infections increase tissue respiration and reallocation of photosynthates before lesions are visible (Daly and others 1961, Verleur 1968). Presumably, secondary infections caused reallocation of photosynthates to infection sites after completion of height growth but before completion of diameter growth. In addition, anthracnose infections would also reduce stored starch normally used for growth the following spring.

The mean defoliation rates in mid-September of year 5 and mid-August of year 6 indicate that approximately half the tagged leaflets had abscised; however, analysis of individual tree responses show that defoliation ranged from 2 to 98 percent with normal distributions in both years. Defoliation percentages in mid-September of year 5 were not correlated with defoliation percentages for mid-August in year 6 for plot means ($r = 0.120$, 71 df), but were correlated using individual tree values ($r = 0.187$, 547 df). Funk and others (1981) also found relatively weak tree-to-tree correlations between years with low incidence of anthracnose with the best correlations occurring between years when there was a high incidence of anthracnose. The weak correlation found between year 5 and year 6 defoliation could also suggest defoliation in year 5 was in response to either the summer drought and/or the low incidence of anthranose found late in the growing season.

Correlation analyses between defoliation percentage and growth increment of the same and following year using either plot means or individual walnut sapling values were generally

stronger for growth increment the following year (table 3). This pattern was strongest for mid-August defoliation of year 6 in response to a high incidence of anthracnose when using individual tree values. Mid-September defoliation in year 5 was highly correlated for growth increment the same year and the following year using either plot means or individual tree values. Height growth in year 6 (38 cm) was approximately 30 percent less than in year 5 (49 cm) suggesting the factors leading to defoliation in year 5, i.e., moisture stress rather than a low incidence of anthracnose, affected growth both years because year 6 had nearly normal precipitation and temperatures.

The relatively weak correlations between defoliation and annual growth increments are in sharp contrast to the strong correlations between tree size and subsequent annual growth increments (table 4). This suggests that the site and environmental factors responsible for accelerated tree growth during the establishment phase, i.e., legume ground covers, continued to strongly affect growth increment and may have masked defoliation response to either anthracnose and/or drought.

To more fully evaluate the effects of premature defoliation, tree size, and their interaction on growth increment, these variables were fitted to a full first and second-order polynomial using regression analysis to generate response surfaces predicting the following year's growth for height, basal diameter, trunk cross-sectional area, and stem volume. Tree size and premature defoliation (mid-September for year 5 and mid-August for year 6) predicted only 37 and 50 percent of the variation for height growth and diameter growth, respectively; however, it predicted nearly 70 percent of the growth in trunk cross-sectional area or stem volume (table 4). Westwood (1978) and Smith and others (1989) indicated that trunk cross-sectional area is a more sensitive measure of tree volume or biomass than are height, stem diameter, or crown area.

Analyses of the response surface generated from the full polynomial confirm that tree size affected future growth more than premature defoliation (figs. 1A to 1D). The coefficient for interaction between tree size and defoliation was significant for both trunk cross-sectional area and stem volume (table 4). Both variables are more closely correlated to tree biomass and demand for photosynthates than are tree height or basal diameter (Westwood 1978). The interaction between tree size and defoliation occurred because small trees showed little response to increasing incidence of defoliation while expected growth on the larger trees showed a linear decrease with increasing incidence of defoliation. This may suggest that the amount of stored photosynthates are limiting tree growth

Table 3.—*Pearson's Correlation Coefficients between premature defoliation percentage and current and following year growth increment using individual tree or plot means for height, basal diameter, trunk cross-sectional area, and stem volume*

		PLOT MEAN BASIS		INDIVIDUAL TREE BASIS	
Growth Increment	Year	Year 5 September defoliation	Year 6 August defoliation	Year 5 September defoliation	Year 6 August defoliation
		— r —	— r –	— r —	— r —
Height	Current	-0.479***	-0.001	-0.274***	-0.044
	Following	-0.417***	-0.177	-0.258***	-0.143***
Basal diameter	Current	-0.449***	-0.113	-0.268***	-0.099*
	Following	-0.376***	-0.179	-0.233***	-0.123**
Trunk area	Current	-0.436***	-0.067	-0.253***	-0.059
	Following	-0.435***	-0.108	-0.241***	-0.078
Stem volume	Current	-0.334**	-0.018	-0.181***	-0.010
	Following	-0.387***	-0.039	-0.207***	-0.042

Correlation coefficients followed by an *, **, or *** are significant across 72 plots or 548 trees at the p = 0.05, 0.01, or 0.001 levels, respectively.

the following year on the better sites while other factors are limiting growth of the small trees usually found in plots without legumes which could be lack of available soil nitrogen on this old field site.

CONCLUSIONS

Black walnut saplings, originally established as seedlings with herbaceous legumes, continued to have more rapid height and diameter growth than walnut saplings growing with resident vegetation dominated by grasses and perennial composites. In addition, saplings with legume ground covers retained their leaflets later into the growing season than saplings with weed vegetation. Premature defoliation of walnut saplings occurred in response to anthracnose and/or drought. Dense stands of a deep-rooted perennial legume accelerated walnut defoliation in response to drought. Decreases in walnut growth were more closely correlated with premature defoliation from the previous growing season than the current growing season.

Models predicting future growth from tree size and premature defoliation indicate that site and environmental factors more strongly control future growth than premature defoliation. Premature defoliation affects the future biomass growth of the faster growing trees more than the growth of the slower-growing trees. Results indicate selection for anthracnose resistance will be an important trait to include in walnut tree improvement programs.

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Table 4.—*Coefficient, significance, and range of values for variables fitted for response surfaces using a full first and second order polynomial to predict following year's growth from tree size and mid-season defoliation*

Variable	COEFFICIENTS FOR RESPONSE SURFACE			
	Height	Diameter	Trunk area	Stem volume
	- cm -	- cm -	- cm^2 -	- cm^3 -
Intercept	10.22394	-0.13906	-0.22082	1487.09**
Size (X)	0.31316***	0.35533***	0.66660***	1.5076***
X-squared (X2)	-0.00011*	-0.01184***	-0.00189***	-0.00001***
Defoliation (Y)	-0.15203	0.00248	0.01938	68.5541***
Y-squared (Y2)	-0.00075	-0.00003	-0.00019	-0.83045***
Interaction (XY)	-0.00049	-0.00045	-0.00125**	-0.00282***
Minimum value	25	0.6	0.4	4
Maximum value	600	12.0	110	40,000
r^2-value	0.37	0.50	0.67	0.68

Coefficients followed by an *, **, or *** are significant (n = 1096) at the p = 0.1, 0.01, or 0.001 levels, respectively.

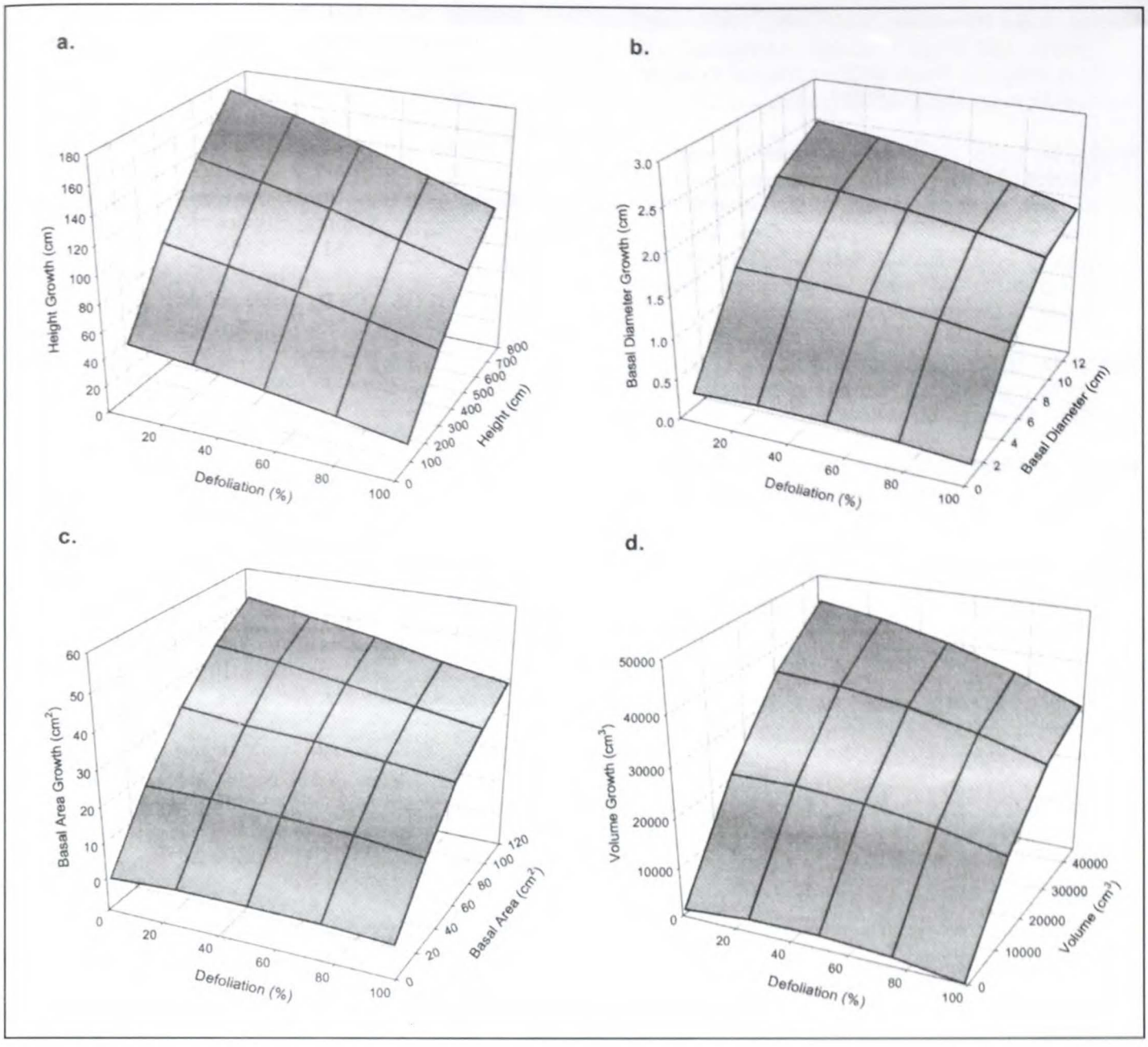

Figure 1.—*Response surface for (a) height, (b) basal diameter, (c) trunk cross-sectional area, and (d) stem volume predicting following year's growth as a function of tree size and amount of late summer or early fall defoliation.*

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APPENDIX I: METRIC TO ENGLISH CONVERSIONS

Convert from:	To:	Multiply by:
bar (10^6 dynes/cm^2)	pound/inch^2 (psi)	14.5
bar (bar)	atmosphere (atm)	0.9869
calorie	British thermal units (Btu)	0.00397
Centigrade (C)	Fahrenheit (F)	1.80C + 32
centimeter (cm)	inch (in)	0.394
centimeters, square (cm^2)	inches, square (in^2)	0.155
centimeter, cubic (cm^3)	inches, cubic (in^3)	0.061
gram (g)	pound (lb)	0.002205
gram (g)	ounces (oz)	0.0353 (for water density)
grams/centimeter, cubic (g/cm^3)		
grams/liter (g/L)	percent (%)	0.100 (in water)
hectare (ha)	acre	2.471
kilogram (kg)	pound (lb)	2.205
kilogram/hectare (kg/ha)	pound/acre	0.893
kilometer (km)	mile (mi)	0.621
kilometer, square (km^2)	miles, square (mi^2)	0.386
kilometer, square (km^2)	acre	247.1
liter (L)	ounce (fluid) (oz)	33.8
liter (L)	quarts (qt)	1.057
liter (L)	feet, cubic (ft^3)	0.0353
liter/hectare (L/ha)	quarts/acre (qt/acre)	0.428
lux (lx)	candle-foot^2 (ft-c)	0.0929
meter (m)	yard (yd)	1.094
meter (m)	feet (ft)	3.281
meter^2 (m^2)	feet^2 (ft^2)	10.76
meter^2/hectare (m^2/ha)	feet^2/acre (ft^2/acre)	4.356 (basal area)
meter^3 (m^3)	feet^3 (ft^3)	35.31
meter^3 (m^3)	board feet, International $^1/_4$" log rule	287.4
meter^3/second (m^3/s)	gallons/second (gal/s)	264 (flow)
meter^3/hectare (m^3/ha)	feet^3/acre (ft^3/acre)	14.29
milligrams/liter (mg/L)	parts per million (ppm)	1.000 (in water)
millimeter (mm)	inch (in)	0.0394
ton (t) or megagram (Mg)	ton (U.S.)	1.102
ton/hectare (t/ha)	ton/acre (t/acre)	0.446

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